Ecological Effects

Appendix E

Introduction

This appendix characterizes the ecological benefits of the Clean Air Act Amendments. Although EPA's focus on a clean environment has long included protection of both ecosystem health and human health, many past analyses, particularly economic analyses, have focused on human health benefits of pollution control. Ecological benefits, by comparison, have not always been as well-represented, for a variety of reasons:

- Ecological impacts may be complex and non-linear, involving relationships at various levels of biological organization. Important ecological effects such as population decline of a keystone species can ripple through a food web and alter community structure and ecosystem function.

- Ecological systems, like human bodies, possess a wide range of adaptive capacities that can mitigate or mask effects and make them difficult to detect. What differentiates, and further complicates the measurement of ecological effects is the lack of sufficient baseline data on natural ecosystem structure and function though successional stages.

- Prevention of ecological effects may be viewed by the public and decision-makers as a lower priority than the protection of human health.

Nonetheless, within the last few decades air pollution started to receive attention for not only affecting human health but also its dramatic injuries to ecosystems. Increased public awareness and research results have led to the development of air pollution research as an important branch of applied biological sciences. Numerous scientific studies have revealed adverse effects of air pollution on natural systems that have, in turn, led to increasingly heightened levels of public concern and subsequent environmental statutory developments. Public policy concerning the regulation of air pollution to mitigate these impacts requires accurate appraisals of the effectiveness of regulatory options, but not until quite recently has it become possible to reliably quantify at least some of the ecological and economic benefits of ecosystem impacts linked to air pollution.

This analysis attempts to incrementally expand the base of quantitative and qualitative information that can be used to assess affects to ecosystems associated with air pollution. There are two major goals of the analysis: to provide a broad overall characterization of the range of effects of air pollutants on ecosystem structure, function, and health; and to extend existing methods and data to characterize the potential magnitude of economic benefits derived from the 1990 Clean Air Act Amendments (CAAA). The economic analysis is focused on a relatively small subset of effects for which ecologists' and economists' understanding of and ability to model an effect is sufficient to develop a quantitative characterization. In most cases, we rely on published, peer-reviewed literature to establish the validity of the methods and data applied.

The remainder of this appendix is comprised of seven major sections. We first provide a broad overview of the ecological impacts of the air pollutants regulated by the CAAA, and then outline the rationale for choosing a subset of these effects for quantitative and economic analysis. Following this largely qualitative characterization of effects, we describe the methods, data, and results used to quantitatively assess benefits of the Clean Air Act Amendments for the following categories of effects:

- Eutrophication of estuaries associated with airborne nitrogen deposition;
- Acidification of freshwater bodies associated with airborne nitrogen and sulfur deposition;
- Reduced tree growth associated with ozone exposure;
- Accumulation of toxics in freshwater fisheries associated with airborne toxics deposition;
- Aesthetic degradation of forests associated with ozone and airborne toxics exposure;
- Other less well-understood effects of air pollution on ecosystem health.

The concluding section includes a summary of those economic estimates that are used in the larger 812 analysis, a summary of major limitations, and recommendations for future research.[1]

Because the breadth and complexity of air pollutant-ecosystem interactions does not allow for comprehensive quantitative analysis of all the ecological benefits of the CAAA we stress the importance of continued consideration of those impacts not valued in this report in policy decision-making and in further technical research. Judging from the geographic breadth and magnitude of the relatively modest subset of impacts that we find sufficiently well-understood to quantify and monetize, it is evident that the economic benefits of the CAAA's reduction of air pollution impacts on ecosystems are of a large magnitude.

Ecological Overview of the Impacts of Air Pollutants Regulated by the CAAA

The purpose of this section is to provide an overview of potential interactions between air pollutants and the natural environment. We identify major single pollutant-environment interactions, as well as the synergistic impacts of ecosystem exposure to multiple air pollutants. Although a wide variety of complex air pollution-environment interactions are described or hypothesized in the literature, for the purposes of this analysis we focus on major aspects of ecosystem-pollutant interactions. We do this by limiting our review according to the following criteria:

- Pollutants regulated by the CAAA;
- Known interactions between pollutants and natural systems as documented in peer-reviewed literature; and,
- Pollutants present in the atmosphere in sufficient amounts after 1970 to cause significant damages to natural systems.

Our understanding of air pollution effects on ecosystems has progressed considerably during the past decades. Previously, air pollution was regarded primarily as a local phenomenon and concern was associated with the vicinity of industrial facilities, power plants or urban areas. The pollutants of concern were gaseous (e.g., sulfur dioxide and ozone) or heavy metals (e.g., lead) and the observed effects were visible stress- specific symptoms of injury (e.g., foliar chlorosis). The most typical approach to document the effects of specific pollutants was a dose-response experiment, where the objective was to develop a regression equation describing the relationship between exposure and some easily measured effect (e.g., growth, yield or mortality). As analytic methods improved and ecology progressed, a broader range of effects of air pollutants were identified and understanding of the mechanisms of effect improved. Observations made on various temporal scales (e.g., long-term studies) and spatial scales (e.g., watershed studies) lead to the recognition

[1] More detailed documentation of the ecological benefits of the CAAA can be found in a series of memoranda and other work products prepared as a part of EPA's benefits analysis and research effort. These memoranda provide comprehensive descriptions of the ecological impacts avoided by the CAAA, methods used to characterize those damages, data sources, and ecological and economic benefits assessments. The more detailed documentation can be obtained through the EPA contacts identified in the Acknowledgements section of the overall report.

that air pollution can affect all organizational levels of biological systems.

In this analysis, we attempt to broadly describe the impacts of air pollutants at all levels of organization, though we are constrained to a review of the most significant impacts on a national scale. For a comprehensive review of the ecological impacts of air pollutants regulated under the CAAA, see *Overview of Ecological Impacts of Air Pollutants Regulated by the 1990 Clean Air Act Amendments* (EPA, 1998a).

Effects of Atmospheric Pollutants on Natural Systems

Ecosystem impacts can be organized by the pollutants of concern and by the level of biological organization at which impacts are directly measured. We attempt to address both dimensions of categorization in this overview. In Table E-1 we summarize the major pollutants of concern, and the documented acute and long-term ecological impacts associated with them. We follow with a description of each of the major pollutant classes and conclude with a summary of pollutant impacts at each level of biological organization.

Table E-1
Classes of Pollutants And Ecological Effects

Pollutant Class	Major Pollutants and Precursors	Acute Effects	Long-term Effects
Acidic deposition	Sulfuric acid, nitric acid Precursors: Sulfur dioxide, nitrogen oxides	Direct toxic effects to plant leaves and aquatic organisms.	Progressive deterioration of soil quality. Chronic acidification of surface waters.
Nitrogen Deposition	Nitrogen compounds (e.g., nitrogen oxides)		Saturation of terrestrial ecosystems with nitrogen. Progressive nitrogen enrichment of coastal estuaries.
Hazardous Air Pollutants (HAPs)	Mercury, dioxins	Direct toxic effects to animals.	Conservation of mercury and dioxins in biogeochemical cycles and accumulation in the food chain.
Ozone	Tropospheric ozone Precursors: Nitrogen oxides and Volatile Organic Compounds (VOCs)	Direct toxic effects to plant leaves.	Alterations of ecosystem wide patterns of energy flow and nutrient cycling.

Acidic Deposition

Acidification is perhaps the best-studied effect of atmospheric pollutant deposition to natural environments. Acidification of ecosystems has been shown to cause direct toxic effects to sensitive organisms as well as long-term changes in ecosystem functions. Acidification can affect all levels of biological organization in both terrestrial and aquatic ecosystems. Adverse effects seen in terrestrial ecosystems can include acute toxic interactions of acids with terrestrial plants or, more importantly, chronic acidification of terrestrial ecosystems leading to nutrient deficiencies in soils, aluminum mobilization, and concomitant decreases in health and biological productivity of forests (Smith, 1990; SOS/T 18, 1989). Similar to terrestrial ecosystems, adverse acidification-induced effects on surface waters may include elevated mortality rates of sensitive species, changes in the composition of communities, and changes in ecosystem-level interactions like nutrient cycling and energy flows. In the United States, acidification-related injuries to aquatic ecosystems may be more significant than injuries of terrestrial sites (EPA, 1995a; NAPAP 1991).

The predominant causes of acidic precipitation are sulfuric and nitric acid (H_2SO_4 and HNO_3). These strong mineral acids are formed from sulfur dioxide (SO_2) and nitrogen oxides (NO_x) in the atmosphere (i.e., they are secondary pollutants). Sulfur compounds are emitted from anthropogenic sources in the form of sulfur dioxide and, to a much lesser extent, primary sulfates, principally from coal and residual-oil combustion and a few industrial processes. Since the late 1960s electric utilities have been the major contributor to SO_2 emissions (NAPAP, 1991 p. 178). The combustion of fuels is the principal anthropogenic source of emissions of NO_x. Such combustion occurs in internal combustion engines, residential and commercial furnaces, industrial boilers, electric utility boilers, engines, and other miscellaneous sources. Because a large portion of anthropogenic NO_x emissions come from transportation sources (i.e., non-point source pollution), NO_x sources are on average more dispersed compared with anthropogenic sources of SO_2 (NAPAP, 1991, p. 189).

In the atmosphere, SO_2 and NO_x are converted to sulfates and nitrates, transported over long distances, and deposited over large areas downwind of point sources or in the vicinity of urban areas. Deposition occurs via three main pathways: 1) precipitation or wet deposition, where material is dissolved in rain or snow; 2) dry deposition, involving direct deposition of gases and particles (aerosols) to any surface; and 3) cloud-water deposition, involving material dissolved in cloud droplets that is deposited when cloud or fog droplets are intercepted by vegetation (NAPAP, 1991, p. 181).

Initially, it was thought that SO_2 emissions were the only significant contributor to acidic deposition. Subsequently, emissions of SO_2 declined substantially between 1970 and 1988 due to a variety of factors, including emissions controls mandated by the Clean Air Act and changes in industrial processes such as the switch of electric utility plants to coal with lower sulfur content (NAPAP, 1991, p. 198). During this period, the role of nitrogen deposition as a contributor to aquatic acidification became apparent. While initial evidence suggested that most deposited nitrogen would be taken up by biota, more recent research has indicated that nitrogen may be leaching from terrestrial systems and causing aquatic acidification.

Comprehensive research on the ecological impacts of acidification is found in the publications of the National Acid Precipitation Assessment Program and EPA's *Acid Deposition Standard Feasibility Study Report to Congress* (1995a). In this analysis we rely heavily upon the extensive research conducted under these two programs.

Nitrogen Deposition

Atmospheric nitrogen deposition to terrestrial and aquatic ecosystems can cause deleterious ecological effects ranging from eutrophication to acidification (as discussed above). Deposition of nitrogen can stimulate nitrogen-uptake by plants and microorganisms and increase biological productivity and growth. Chronic deposition of nitrogen may adversely affect biogeochemical cycles of watersheds by progressively saturating terrestrial portions with nitrogen. Nitrogen saturation is a gradually occurring process, during which watersheds undergo progressive changes in their nitrogen cycle. This process can lead to increases in the amount of nitrogen leached into lower-elevation terrestrial ecosystems, wetlands and, most notably, surface waters (Stoddard, 1994; Aber et al., 1989).

Among the most pernicious effects of nitrogen leaching from terrestrial ecosystems can be acidification of fresh water bodies (as previously discussed) and eutrophication of estuaries (Richardson, 1996; Vollweider et al., 1990). Similar to terrestrial ecosystems, nitrogen enrichment of coastal estuaries can have a fertilizing effect, stimulating productivity of algae, marine plants (Vitusek and Howarth, 1991) and aquatic animals, including fish and shell fish. If eutrophication is excessive, however, it is likely to result in serious damages to estuarine ecosystems. Specifically, massive algae blooms can develop, leading to declining oxygen levels, habitat loss, and declines in fish and shellfish populations.

Nitrogen loading to estuaries is a major and growing problem. A 1996 inventory of estuarine water quality performed by coastal states and encompassing 72 percent of estuaries in the U.S. shows that nutrient enrichment pollutes 6,254 square miles (22 percent) of the surveyed waters, and contributes to 57 percent of all the reported water quality problems. At a recent meeting of National Estuary Program directors, eleven out of twenty-eight directors ranked nutrient overloading as a high priority issue for their programs, and seven additional directors ranked it as a mid-level priority (EPA, 1997b). Eighty-six percent of East Coast estuaries are considered susceptible to nitrogen enrichment (EPA, 1997c), and many coastal communities are finding that the nutrient loading problem is already so severe that they must add advanced wastewater treatment to existing plants, add infrastructure to promote water reuse, and impose stricter controls on all development and agricultural practices (EPA, 1997a).

Atmospherically derived nitrogen makes up a sizable fraction of total nitrogen inputs to estuaries in the Northeast (Hinga et al. 1991; Jaworski et al. 1997; Paerl 1997; Paerl et al. 1990; McMahon and Woodside, 1996; Rendell et al. 1993; Valiela et al. 1997). Atmospheric nitrogen is deposited to waters and watersheds in wet (rain, snow, and fog) and dry (aerosols and gases) forms. Approximately 10 to 50 percent of total nitrogen load to coastal waters is derived from direct and indirect atmospheric deposition. Estuaries on the eastern seaboard, and those downwind of urban areas tend to have a larger percentage of total nitrogen coming from atmospheric deposition.

Hazardous Air Pollutant Deposition

Hazardous air pollutants (HAPs), are a general category of toxic substances covered under a single title of the Clean Air Act. Title III lists 189 HAPs, though only five are responsible for the majority of currently documented ecosystem impacts. These HAPs are mercury, polychlorinated biphenyls (PCBs), chlordane, dioxins, and dichlorodiphenyl-trichloroethane (DDT). The use of three of these compounds (PCBs, chlordane, and DDT) was effectively illegal in the United States prior to 1990 (EPA 1992), and there are currently no plans for additional CAAA regulations of these compounds (Federal Register Unified Agenda 1998). Emissions of the remaining two toxins, mercury and dioxins, continue to cause ecosystem impacts.

Mercury (Hg) is a toxic element found ubiquitously throughout the environment. Unlike many HAPs, much of the mercury released to the environment comes from natural sources. Anthropogenic sources can also release mercury to the environment. Estimates of the percentage of mercury emissions attributable to anthropogenic sources range from 10 to 80 percent (Mason et al. 1994, Hudson et al. 1995, Stein et al. 1996), although most estimates cluster between 40 and 75 percent (EPA 1997d).

About 80 percent of all anthropogenic mercury loadings to the environment are from air emissions. Global atmospheric concentrations of mercury have approximately tripled since pre-industrial times (Mason et al. 1994). Atmospheric deposition of mercury has increased by a factor of about 3.7 (Swain et al. 1992), and the concentration of mercury in sediments in remote lakes has increased by a factor of 2.3 (Lucotte et al. 1995). These findings suggest that approximately 57 to 73 percent of atmospherically deposited mercury is anthropogenic in origin.

Atmospheric deposition of mercury and its subsequent movement in ecosystems may result in the concentration of mercury within organisms ("bioaccumulation") and its subsequent transfer throughout the food chain. As a consequence, mercury tends to accumulate along the hierarchical organization of food webs, with increasing concentrations found in animals at higher levels of the food chain ("biomagnification"). Fish, birds and mammals are among the group of organisms most threatened by mercury contamination of the environment. Symptoms may range from behavioral abnormalities to reduced reproductive success and death (EPA, 1997d). In 1996, mercury levels in fish were high enough that 11 states had mercury-based statewide fish consumption advisories. Twenty-eight more had at least one water body under advisory because of mercury concerns. These observations suggest that atmospheric mercury deposition may contribute significantly to mercury levels in freshwater ecosystems nationally.

Mercury is a neurotoxin that, at sufficient levels, can cause neurologic damage and death in both animals and humans. Adverse effects on wildlife include neurotoxicity, reproductive, and developmental effects (EPA 1997d). While fish are unlikely to experience toxic effects from mercury poisoning in the absence of point discharges, piscivorous predators and predators who eat piscivores accumulate more mercury and may suffer from mercury poisoning. However, the only species for which there is currently strong evidence of poisoning from atmospheric mercury are the common loon and possibly the Florida panther. It is unclear whether other piscivorous species, such as kingfishers, mink, and river otters, have suffered adverse health effects as a consequence of atmospheric mercury deposition (EPA 1997d).

Mercury is likely to persist at levels of concern in ecosystems for some time. The majority of atmospherically-released mercury is deposited to terrestrial environments, where it is largely sequestered. However, as mercury accumulates in soils, some amount (less than 30 percent of that which is deposited within a watershed) will be slowly released to freshwater bodies and oceans. Modeling efforts by Swain et al. (1992, reviewed in Mason et al. 1994) suggest that the retention of mercury by some lakes is essentially complete. Studies by Mason et al. (1994) predicted that elimination of anthropogenic mercury presently in the oceans and in the atmosphere would take 15 to 20 years after the complete termination of all anthropogenic emissions. Because of mercury's persistence in terrestrial and aquatic environments, it will probably take some time for reductions in mercury emissions to be notable in ecosystems (Swain et al. 1992, reviewed in Mason et al. 1994).

The other HAPs of concern are polychlorinated dibenzo-p-dioxins (PCDDs), a group of 75 organochlorine compounds that are sometimes referred to as dioxins. The most toxic member of this group is 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Because TCDD is the most toxic dioxin, the toxicity of a dioxin mixture is often expressed as the *toxic equivalency* (TEQ) of some amount of TCDD. Polychlorinated dibenzofurans (PCDFs) are close chemical relatives of PCDDs. Both classes of compounds are produced by the same processes, and both are ubiquitous in the environment (WHO 1989). TEQ estimates are often given jointly for dioxins and furans.

Dioxins and furans, unlike mercury, are not natural to the environment. They are formed during the combustion of wastes and fossil fuels, and as a consequence of fires or spills that involve particular chemicals like benzenes or PCBs. They are also formed as by-products in both the manufacturing of other chemicals and in pulp and paper mill bleaching processes (WHO 1989, EPA 1992a). EPA estimates that combustion sources emit over ten times as many TEQs as did all other categories combined.

Dioxins accumulate in the fat of animals and bioaccumulate through food chains. Fish are among the most sensitive vertebrates to the effects of TCDD, especially during early life stages. Fish are exposed to dioxins primarily through their food (EPA 1993), but some studies have reported that they can also absorb the trace amounts of dioxins present in water. EPA (1992) reported bioconcentration factors (BCFs)[2] for dioxins in fish of 5,000 to 9,000, and EPA (1993) estimated that bioaccumulation factors (BAFs) for lake trout can be on the order of 500,000 to 1,200,000.[3] Toxic effects on young fish include decreased feeding, weight loss, and fin necrosis; however, with a few exceptions, TCDD levels in the environment are generally too low to result in toxicity to juvenile or adult fish (EPA 1993, Walker and Peterson 1994).

The risk that dioxins pose to other wildlife is difficult to assess because both laboratory and field studies in this area are limited (EPA 1993, Giesey et al. 1994). One study (White et al. 1994) found that wood duck eggs from a contaminated area had levels of PCDDs and PCDFs 50 times higher than levels in control eggs. The contaminated nests were significantly less successful than control nests, and contaminated ducklings also suffered from teratogenic effects.

TCDD is an extremely stable chemical and is unlikely to be significantly degraded by chemical or biological hydrolysis under normal environmental conditions. Its half-life in soils may be on the order of a decade or more, and it may be even more persistent in aquatic sediments (Webster and Commoner 1990). For example, Johnson et al. (1996) found that, though TCDD levels in fish and sediments from an Arkansas river declined significantly during the 12 years following the initial pollution of the river, fish from some locations continued to have levels of TCDD that exceeded Food and Drug Administration (FDA) guidelines. TCDD is subject to photochemical degradation, but since the penetration of light into soils and many natural water bodies is limited, this degradation is not likely to be environmentally significant (WHO 1989, Zook and Rappe 1990). Because of dioxins' toxicity and persistence, their presence in freshwater ecosystems is likely to be an issue of concern for decades.

Tropospheric Ozone

Ozone pollution is widespread in the eastern United States, in southern California, and in the vicinity of most major cities. Many of the observed effects of ozone on vegetation are related to direct toxic or harmful interactions with essential physiological functions of plants and subsequent reductions in biomass production (reduced growth). Damages to plants are commonly manifested as stress specific symptoms such as necrotic spots of plant leaves, acceleration of leaf aging, and reduced photosynthesis. Ozone damages at the community and ecosystem-level vary widely depending upon numerous factors, including concentration and temporal variation of tropospheric ozone, species composition, soil properties and climatic factors. In most instances, responses to chronic or recurrent exposure are subtle and not observable for many years. These injuries can cause stand-level forest decline in sensitive ecosystems (EPA, 1996; McBride et al., 1985; Miller et al., 1982).

Species that are particularly sensitive to ozone can be found among all groups of plants. Although many visible injuries have occurred in conifer species (e.g., Ponderosa and Jeffrey pine), a variety of deciduous trees and shrubs are also sensitive to ozone. Black cherry, many poplars (genus *Populus*) and many fruit trees including almond (*Prunus amygdalis* Batsch), peach (*Prunus persica*), and plum (*Prunus domestica*) trees are all

[2]Bioconcentration factors (BCFs) are calculated based on laboratory experiments. BCFs represent the ratio between the chemical's concentration in the organism to its concentration in the water, but unlike bioaccumulation factors (BAFs), they measure only how much of a chemical an organism accumulates as a consequence of its exposure to contaminated water. BCFs do not measure contaminant uptake as a function of exposure to contaminated food or sediments (EPA 1993).

[3]Because dioxins have such low solubility, accurately measuring their concentrations in water is extremely difficult. For this reason, any reported BCFs and BAFs should be examined carefully.

affected by elevated levels of ozone (EPA 1996a). In annual species, effects of ozone on production occur through changes in allocation of carbohydrates and can result in reduced seed production. Many annual plant species, including commercial crops, are among the most sensitive species. The National Crop Loss Assessment Network (NCLAN 1988), a multi-year program of the EPA, established that ambient ozone levels cause physical damages to crop plants and statistically significant reductions in agricultural yields.

Ecosystems with known damages that are attributed to ozone include the San Bernardino Mountains of Southern California, the Sierra Nevada Mountains, and sites in the vicinity of urban areas throughout the country. According to EPA (1996a), the San Bernardino Mountain range is by far the most severely ozone-impacted ecosystem. This mixed-conifer forest ecosystem has been exposed to chronically elevated ozone levels over a period of 50 or more years. This exposure has resulted in major changes of ecosystem characteristics, including species composition, nutrient cycling and energy flow. The first indications of ozone damages to the ecosystem were observed on the more sensitive members of the forest community: individual Ponderosa and Jeffrey pines. Direct injuries included visible foliar damage, premature needle senescence, reduced photosynthesis, altered carbon allocation, and reduction of growth rates and reproductive success. Changes in the energy available to trees (i.e., changes in carbohydrate production and allocation) influenced interactions with predators, pathogens and symbionts. Subsequently, the accumulation of weakened trees resulted in heavy bark beetle attack that significantly elevated mortality rates and extensive salvage logging during the 1960s and 1970s (Miller and McBride, 1998). Alterations in the composition and population density of the fungal microflora weakened soil microbial organisms and slowed the rate of decomposition, leading to the accumulation of a thick needle layer under stands with the most severe needle injuries and defoliation. Reduced production of seeds and fruits also affected the amount of food available to small vertebrates in the ecosystem, thereby affecting the local food chain (EPA 1996a). Similarly, ozone concentrations capable of causing injury to the Sierra Nevada Mountains have been occurring for many years, but injury to sensitive trees has never reached the same proportions as in the San Bernardino Forest. Significant differences in both the forest stand composition (e.g., the presence of fewer conifers and more hardwoods), and site dynamics have probably played an important role in determining the different ecosystem responses.

In each of these areas, ozone may act synergistically with other stress factors to induce further damages to vegetation. In the eastern United States, for example, regionally elevated levels of tropospheric ozone co-occur with high deposition rates of nitrogen, sulfur and acids. These multiple stress factors may have acted synergistically in injuring many high elevation forests throughout the eastern United States.

Multiple Stresses and Patterns of Exposure

Although air pollutants can be grouped into classes according to their effects, as described above, it is recognized that one pollutant (or one class of pollutants) does not solely impact most ecosystems. Many environmental damages are the result of the combined action of multiple stress factors, including several types of air pollution and other anthropogenic or natural stress factors.

The recognition of interactions between several types of pollutants and between pollutants and other kinds of stress has introduced a new level of complexity in air pollution research. In many cases, various stress factors act synergistically to induce damages to ecosystems. These multiple stress factors can include: (1) various kinds of air pollutants (e.g., acidic deposition, ozone and nitrogen deposition); (2) other anthropogenic stress factors such as harvesting, overfishing or habitat disruption (e.g., the disruption of ecosystems by roads or urban areas); (3) environmental factors including availability of water, nutrients, light, or temperature (including heat and frost); and (4) biological factors such as animals feeding on plants, pathogens, and the status of micro-organisms in the soil (Taylor et al., 1994;

Winner, 1994; Smith, 1990). In the eastern US, for example, elevated deposition rates of nitrogen, sulfur and acids co-occur with regionally elevated levels of tropospheric ozone and climatic stress factors.

In addition, air pollutants have indirect effects that are at least as important as direct toxic effects on living organisms. Indirect effects include those in which the pollutants(s) alter the physical or chemical environment (e.g., soil properties) the plant's ability to compete for limited resources (e.g., water, light), or its ability to withstand pests or pathogens. Examples are excessive availability of nitrogen, soil depletion caused by acidic deposition, and changes in the ability to adapt to cold temperatures induced by acidic deposition (Taylor et al., 1994; NAPAP, 1991). Unfortunately, few mechanisms of interactions between various stress factors are known, and interpretations of scientific findings are usually associated with a high degree of uncertainty.

The situation is further complicated by the fact that the specific temporal and spatial patterns of pollutant exposures play a significant role in the response of organisms and ecosystems to air pollution. Temporal patterns include timing, duration and patterns of recurrent exposure to a specific pollutant or pollutants. For example, plant response to peak concentrations of ozone during daylight can be more severe (compared to exposure to the same level of ozone at night) because uptake of ozone is often higher during the day (EPA, 1996a). Spatial patterns include proximity of ecosystems to various pollution sources and the identification of specific source-receptor relationships between ecosystems and pollution sources (Taylor et al., 1994). For example, the deposition of most pollutants occurs after long-range atmospheric transport, with deposition rates depending upon climate, land-use and geology.

Summary of Ecological Impacts from Air Pollutants Regulated by the CAAA

We summarize major examples of air pollution interactions with various levels of biological organization in Table E-2 through E-4. We organize these interactions according to classes of pollutants and injuries they cause, the various levels of biological systems, and types of affected ecosystems. It is important to note that interactions listed are intended to illustrate the range of possible adverse effects. These effects are examples for a wide variety of interactions but do not cover all aspects of air pollution-environment interactions.

Table E-2
Interactions Between Acid Deposition and Natural Systems
At Various Levels of Organization

		Examples of Interactions	
Spatial Scale	**Type of Interaction**	**Acidification of Forests**	**Acidification of Streams and Lakes**
Molecular and cellular	Chemical and biochemical processes	Damages to epidermal layers and cells of plants through deposition of acids.	Decreases in pH and increases in aluminum ions cause pathological changes in structure of gill tissue in fish.
Individual	Direct physiological response	Increased loss of nutrients via foliar leaching.	Hydrogen and aluminum ions in the water column impair regulation of body ions.
	Indirect effects: Death due to ionoregulatory failure. Acidification can indirectly affect response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress.	Cation depletion in the soil causes nutrient deficiencies in plants. Concentrations of aluminum ions in soils can reach phytotoxic levels. Increased sensitivity to other stress factors like pathogens and frost.	Aluminum ions in the water column can be toxic to many aquatic organisms through impairment of gill regulation. Acidification can indirectly affect submerged plant species, because it reduces the availability of dissolved carbon dioxide (CO_2).
Population	Change of population characteristics like productivity or mortality rates.	Decrease of biological productivity of sensitive organisms. Selection for less sensitive individuals. Microevolution of resistance.	Decrease of biological productivity of sensitive organisms. Selection for less sensitive individuals. Microevolution of resistance.
Community	Changes of community structure and competitive patterns	Alteration of competitive patterns. Selective advantage for acid-resistant species. Loss of acid sensitive species and individuals. Decrease in productivity. Decrease of species richness and diversity.	Alteration of competitive patterns. Selective advantage for acid-resistant species. Loss of acid sensitive species and individuals. Decrease in productivity. Decrease of species richness and diversity.
Local Ecosystem (e.g., landscape element)	Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc.	Progressive depletion of nutrient cations in the soil. Increase in the concentration of mobile aluminum ions in the soil.	Measurable declines of decomposition of some forms of organic matter, potentially resulting in decreased rates of nutrient cycling.
Regional Ecosystem (e.g., watershed)	Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity.	Leaching of sulfate, nitrate, aluminum, and calcium to streams and lakes. Acidification of aquatic bodies.	Additional acidification of aquatic systems through processes in terrestrial sites within the watershed.

Table E-3
Interactions Between Nitrogen Deposition and Natural Systems At Various Levels of Organization

		Examples of Interactions	
Spatial Scale	**Type of Interaction**	**Eutrophication and Nitrogen Saturation of Terrestrial Landscapes**	**Eutrophication of Coastal Estuaries**
Molecular and cellular	Chemical and biochemical processes	Assimilation of nitrogen by plants and microorganisms	Assimilation of nitrogen by plants and microorganisms.
Individual	Direct physiological response.	Increases in leaf- size of terrestrial plants.	Increase in growth of marine plants.
	Indirect effects: Response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress.	Decreased resistance to biotic and abiotic stress factors like pathogens and frost. Disruption of plant-symbiont relationships with mycorrhiza fungi.	Injuries to marine fauna through oxygen depletion of the environment. Loss of physical habitat due to loss of sea-grass beds. Injury through increased shading. Toxic blooms of plankton.
Population	Change of population characteristics like productivity or mortality rates.	Increase in biological productivity and growth rates of some species.	Increase in biological productivity. Increase of growth rates (esp. of algae and marine plants).
Community	Changes of community structure and competitive patterns	Alteration of competitive patterns. Selective advantage for fast growing species and individuals that efficiently use additional nitrogen. Loss of species adapted to nitrogen-poor environments.	Excessive algal growth. Changes in species composition. Decrease in sea-grass beds.
Local Ecosystem (e.g., landscape element)	Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc.	Magnification of the biogeochemical nitrogen cycle. Progressive saturation of microorganisms, soils, and plants with nitrogen.	Magnification of the nitrogen cycle. Depletion of oxygen, increased shading through algal growth.
Regional Ecosystem (e.g., watershed)	Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity.	Leaching of nitrogen from terrestrial sites to streams and lakes. Acidification of aquatic bodies. Eutrophication of estuaries.	Additional input of nitrogen from nitrogen-saturated terrestrial sites within the watershed.

Note: See *Overview of Ecological Impacts of Air Pollutants Regulated by the 1990 Clean Air Act Amendments* (IEc 1998) for sources.

Table E-4
Interactions of Mercury and Ozone with Natural Systems At Various Levels of Organization

Spatial Scale	Type of Interaction	Examples of Interactions: Mercury in streams and lakes	Examples of Interactions: Ozone
Molecular and cellular	Chemical and biochemical processes	Mercury enters the body of vertebrates and binds to sulfhydril groups (i.e. proteins).	Oxidation of enzymes of plants. Disruption of the membrane potential.
Individual	Direct physiological response.	Neurological effects in vertebrates. Behavioral abnormalities. Damages to the liver.	Direct injuries include visible foliar damage, premature needle senescence, reduced photosynthesis, altered carbon allocation, and reduction of growth rates and reproductive success.
	Indirect effects: Response to altered environmental factors or alterations of the individual's ability to cope with other kinds of stress.	Few interactions known. Damages through increased sensitivity to other environmental stress factors could occur, for example, through impairment of immune response.	Increased sensitivity to biotic and abiotic stress factors like pathogens and frost. Disruption of plant-symbiont relationship (mychorrhizae), and symbionts.
Population	Change of population characteristics like productivity or mortality rates.	Reduced reproductive success of fish and bird species. Increased mortality rates, especially in earlier life stages.	Reduced biological productivity. Selection for less sensitive individuals. Possibly microevolution for ozone resistance.
Community	Changes of community structure and competitive patterns	Loss of species diversity of benthic invertebrates.	Alteration of competitive patterns. Selective advantage for ozone-resistant species. Loss of ozone sensitive species and individuals. Reduction in productivity.
Local Ecosystem (e.g.,landscape element)	Changes in nutrient cycle, hydrological cycle, and energy flow of lakes, wetlands, forests, grasslands, etc.	Not well understood.	Alterations of ecosystem-wide patterns of energy flow and nutrient cycling.
Regional Ecosystem (e.g., watershed)	Biogeochemical cycles within a watershed. Region-wide alterations of biodiversity.	Not well understood.	Region-wide loss of sensitive species.

Note: See *Overview of Ecological Impacts of Air Pollutants Regulated by the 1990 Clean Air Act Amendments* (IEc 1998) for sources.

Predicting ecological impacts of air pollution at the regional scale or for the United States as a whole would require an understanding of interactions at many temporal and spatial scales, where there is currently a general lack of data. Furthermore, there is limited transferability of existing information between various spatial and temporal scales and between geographic regions. However, we can reach several general conclusions, based on the existing literature.

- Although ambient concentrations of most air pollutants significantly decreased after the Clean Air Act of 1970, some pollutants still occur in concentrations high enough to directly injure living organisms. These direct injuries can be observed, for example, in areas with high ambient levels of tropospheric ozone or in some high-elevation ecosystems that are exposed to high levels of acid deposition (EPA, 1996a; NAPAP, 1991).

- Air pollutants have indirect effects that are at least as important as direct toxic effects on living organisms. Indirect effects include those in which the pollutant alters the physical or chemical environment (e.g., soil properties), the plant's ability to compete for limited resources (e.g., water, light), or the plant's ability to withstand pests or pathogens. Examples are excessive availability of nitrogen, depletion of nutrient cations in the soil by acid deposition, mobilization of toxic elements such as aluminum, and changes in winter hardiness (Taylor et al., 1994). As is true for other complex interactions, indirect effects are more difficult to observe than direct toxic relationships between air pollutants and biota, and there may be a variety of interactions that have not yet been detected.

- There is a group of pollutants that tend to be conserved in the landscape after they have been deposited to ecosystems. These conserved pollutants are transformed through biotic and abiotic processes within ecosystems, and accumulate in biogeochemical cycles. These pollutants include, but are not limited to, hydrogen ions (H+), sulfur (S) and nitrogen (N) containing substances, and mercury (Hg). Chronic deposition of these pollutants, can result in progressive increases in concentrations and cause injuries due to cumulative effects. Indirect, cumulative damages caused by chronic exposure (i.e., long-term, moderate concentrations) to these pollutants may increase in magnitude over time frames of decades or centuries with very subtle annual increments of change. Examples are N-saturation of terrestrial ecosystems, cation depletion of terrestrial ecosystems, acidification of streams and lakes, and accumulation of mercury in aquatic food webs (Pitelka 1994; Taylor et al. 1994; Likens et al. 1996; EPA 1997e).

- Damages to ecosystems are most likely caused by a combination of environmental stress factors with every interactive stress or else have a mechanistic model that incorporates interactions among pollutants. Unfortunately neither approach is yet possible. These include anthropogenic factors such as air pollution and other environmental stress factors such as low temperature, excess or limited water, and limited availability of nutrients. The specific combinations of factors differ among regions and ecosystems where declines have been observed (Taylor et al., 1994; Winner, 1994; Smith, 1990). To accurately predict the impacts of multiple acting stress factors we would have to build a catalogue of research results that defines the response of every plant species to every air pollutant, with every interactive stress or else have a mechanistic model that incorporates interactions among pollutants. Unfortunately neither approach is yet possible.

- Pollutant-environment interactions are complicated by the fact that biotic and abiotic factors in ecosystems change dramatically over time. Besides oscillations on a daily basis, and changes in a seasonal rhythm, long range successional changes occur over time periods of years, decades, or even centuries. These temporal variations occur in polluted and pristine ecosystems, and no single point in time or space can be defined as representative of the entire system.

Long-term impacts of air pollution are often manifested in interactions at the regional scale. The history of lead pollution may provide a useful illustration of impacts of air pollution long after deposition rates have declined significantly due to environmental regulations. Historically, scientists were concerned about lead deposition because of its high affinity to soil organic matter and its accumulation in the litter layers of soils. Starting around 1960, lead accumulated in forest soils in the northeastern United States as a result of human activities. Following a significant decline of combustion of leaded gasoline between 1970 and 1988, deposition rates dropped, and decreases in lead levels in soils and rivers have been observed throughout the United States. Apparently forest floors have responded rapidly to the decline of lead input, and instead of accumulating lead, forest soils are now slowly releasing lead to the underlying mineral horizon. It has been estimated that sometime in the middle of the next century, forests will begin to release anthropogenic lead deposited after 1960 to rivers and streams (Miller and Friedland, 1994), where it may cause unforeseen damages to aquatic ecosystems.

There is evidence that current air pollution is an important environmental stress factor over large areas of the United States and other countries, even if effects have not yet been fully documented. Actions taken now to reduce air emissions may have consequences far into the future and may affect ecosystems in ways that are not yet known. Because it is not yet possible to predict what long-term, continuous exposure to multiple pollutants might do to ecosystem structure and function, it may be concluded that the ecological benefits of air pollution control lie in the prevention of long-term damages to resources and the potential for increased recovery rates, as well as the more traditional prevention of acute injuries. Because it is not yet possible to predict what long-term, continuous exposure to multiple pollutants might do to ecosystem structure and function, it may be prudent to focus on the prevention of possible long-term damages to resources and preserve the potential for increased recovery rates, as well as preventing more traditional acute injures to ecosystems.

Methodological Overview

In this section we describe the methods for characterizing the economic benefits of reducing several classes of ecological impacts through the regulations of the CAAA. As indicated in the previous section, it is not possible to characterize and quantify all impacts associated with air pollution. Instead, we select those impacts amenable to quantitative analysis, using two criteria:

Criterion #1: The endpoint must be an identifiable service flow

Criterion #2: A defensible link must exist between changes in air pollution emissions and the quality or quantity of the ecological service flow, and quantitative models must be available to monetize these changes

The use of these criteria greatly constrains the range of impacts that can be treated in this analysis. While the previous section identifies many pollutant-ecosystem interactions, only a handful are understood and have been modeled to an extent sufficient to reliably quantify their impact. We attempt to present both reliable quantitative information regarding the benefits of the CAAA while demonstrating the potential magnitude of the ecological benefits of the CAAA if all impacts were valued. A more detailed description of the choice of endpoints is found in

Methods for Selecting Monetizable Benefits Derived from Ecological Resources as a Result of Air Quality Improvements Attributable to the 1990 Clean Air Act Amendments, 1990-2010 (IEc 1998b) and Characterizing Economic Benefits of Reducing Impacts to Ecosystem Integrity (IEc 1998c).

Using Service Flow Endpoints for Valuation

The theoretical basis of economic benefits assessment is that ecosystems provide services to humankind, and that those services have economic value. The application of this theory requires the isolation of service flows that have market values or are otherwise amenable to available methods for determining value in the absence of formal markets. Freeman (1997) provides one possible grouping of ecological service flows:

- Sources of material inputs to the economy, including fossil fuels, wood products, minerals, water, and animals;
- Life support services, including breathable air and a livable climate;
- Amenities that provide opportunities for active recreation and passive enjoyment of nature, including nonuse values; and
- Processing of waste products that are generated by economic activity and discharged into the environment.

Available methods do not exist to comprehensively value each of these service flows for all ecosystems. Generally, we are limited to those service flows that either are sources of material inputs or natural amenities that involve active recreation. Impacts to these service flows that can be valued tend to manifest themselves immediately and can be readily measured and assessed in terms of the proven cause and effect relationships. The result is that we can value only a small subset of the ecosystem benefits from environmental regulations in an analysis of national scope.

Based on the constraints of economic valuation methods and data, we select from the host of ecosystem impacts identified in the previous section a set of service flows as candidate endpoints for analysis. These endpoints are listed in Table E-5.

Table E-5
Ecological Impacts with Identifiable Human Service Flows

Pollutant Class	Ecosystem Effect	Service Flow Impacted
Acidification (H_2SO_4, HNO_3)	High-elevation forest acidification resulting in dieback	Forest aesthetics
	Freshwater acidification resulting in aquatic organism (e.g. fish) population decline	Recreational fishing
	Changes in biological diversity and species mix in terrestrial and aquatic systems	Existence value for maintenance of biological diversity
Nitrogen Saturation and Eutrophication (NO_x)	Freshwater acidification resulting in aquatic organism (e.g. fish) population decline	Recreational fishing
	Estuarine eutrophication causing oxygen depletion and changes in nutrient cycling	Recreational and commercial fishing
	Changes in biological diversity and species mix in terrestrial and aquatic systems	Existence value for maintenance of biological diversity
Toxics Deposition (Mercury, Dioxin)	Terrestrial bioaccumulation of mercury and dioxin	Hunting, wildlife aesthetics
	Aquatic bioaccumulation of mercury and dioxin	Recreational and commercial fishing
	Changes in biological diversity and species mix in terrestrial and aquatic systems	Existence value for maintenance of biological diversity
Tropospheric Ozone (O_3)	Terrestrial plant foliar damage causing lower productivity and reduced competitiveness	Commercial timber productivity, forest aesthetics, existence value
Multiple Pollutant Stress	Ecosystem deterioration resulting in visual effects, habitat loss, and changes in biological diversity and species mix caused by synergistic action of several pollutants	Ecosystem aesthetics, ecosystem existence value

Defensible Links and Quantitative Modeling Requirements

The second criterion for endpoint selection is satisfied when complete data and model coverage is available to describe the impacts of air pollutants. We briefly describe the types of data and models necessary to accomplish quantitative benefits assessment, then identify those endpoints that we can pursue in this analysis.

In order to determine changes in ecological service flows, defensible links between pollution emissions and service flow changes must be quantitatively modeled. Described generally, five steps are necessary to complete a quantitative analysis; emissions characterization; environmental fate and transport assessment; exposure characterization; ecosystem effects characterization; and economic behavior models.

Emissions characterization requires models that project the level of air pollutants entering the atmosphere over the period of time in question for both factual and counterfactual scenarios under consideration in the analysis. In our analyses the factual scenario is the level of emissions in the United States generated between 1990 and 2010, as regulated by the CAAA (Post-CAAA). The counterfactual scenario is the level of emissions during the same period without the regulations promulgated under the CAAA (Pre-CAAA)[4].

The geographic transport and deposition of air pollutants are estimated using models that consider multiple chemical and meteorologic factors. The section 812 prospective analysis of the CAAA uses three models, detailed in Appendix C and *Air Quality Modeling to Support the Section 812 Prospective Analysis* (prepared for EPA by Systems Applications International, Inc., 1999)[5].

In cases where the presence of a pollutant in a geographic region, as estimated by dispersion models, is not an adequate measure of the exposure of biota to the pollutant, an exposure model is required. These models must take biotic and abiotic ecosystems processes into account.

Once the exposure of the biota in question is estimated, the physiological effect of that exposure must be estimated. Dose-response functions that describe the effects of varying levels of pollutants to specific organisms are derived from laboratory, field, and modeling experiments. The intensive nature of this research and the necessity of studying each species individually causes this link to be weak in most quantitative ecological assessments.

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[4]For all analyses in this report, emissions under each scenario are based upon EPA's National Emissions Inventory (NEI) with modeling provided by the Emissions Reduction and Cost Analysis Model (ERCAM). See Appendix A for details.

[5]The three regional-scale air quality modeling systems applied include the regulatory Modeling System for Aerosols and Deposition (REMSAD), the Regional Acid Deposition Model (RADM), and the Urban Airshed Model (UAM-IV and UAM-V). In addition, this prospective ecological benefits assessment uses results from the Regional Lagrangian Model of Air Pollution (RELMAP) to estimate mercury and dioxin deposition.

Table E-6
Model Coverage for Candidate Endpoints for Quantitative Assessment

Pollutant	Endpoint	Emissions Model	Transport and Deposition	Exposure Model	Dose-response Functions	Economic Model
Acidification (H_2SO_4, HNO_3)	Forest aesthetics	NEI, ERCAM	RADM	Not Required	Multiple available	Only site-specific models available
	Recreational fishing	NEI, ERCAM	RADM	MAGIC (region specific)	Multiple available	Only site-specific models available
	Biological diversity existence value	NEI, ERCAM	RADM	MAGIC (region specific)	Multiple available	Only site-specific models available
Nitrogen Saturation and Eutrophication (NO_x)	Recreational and commercial fisheries	NEI, ERCAM	RADM	Estuary-specific models available	None Available	Only site-specific models available
	Biological diversity existence value	NEI, ERCAM	RADM	Estuary-specific models available	Multiple available	None Available
Toxics Deposition (Hg, Dioxin)	Forest aesthetics	None Available	RELMAP, ISC3	None Available	Multiple available	Only site-specific models available
	Hunting, wildlife aesthetics	None Available	RELMAP, ISC3	None Available	Multiple available	Only site-specific models available
	Recreational and commercial fishing	None Available	RELMAP, ISC3	IEM-2M (site specific)	Multiple available, or consumption advisory limits can be used	Only site-specific models available
	Biological diversity existence value	None Available	RELMAP	None Available	Multiple available	None Available
Multiple Pollutant Stress	Ecosystem aesthetics, ecosystem existence value.	NEI, ERCAM		None Available	None Available	None Available

NEI: National Emissions Inventory; ERCAM: Emissions Reduction and Cost Analysis Model; RADM: Regional Acid Deposition Model;REMSAD: Regulatory Modeling System for Aerosols and Deposition; RELMAP: Regional Lagrangian Model of Air Pollution; UAM: Urban Airshed Model; TAMM: Timber Assessment Market Model, developed and maintained by the U.S. Forest Service.

When pollutant doses are sufficiently high to cause physiologic responses in biota, ecological service flows may be affected. In order to monetize these impacts, a model of the economic behavior associated with the service flow must be developed. Economic models are specific to the service flow and the consuming population, and not all service flows have adequate economic models that describe their value. For example, recreational fishing models account for the preferences and geographic distribution of anglers as well as the site characteristics of the fisheries. These data are site specific, making the model specification fairly non-transferable.

Table E-6 describes the extent to which models are available to estimate the full chain of defensible links for the ecological endpoints identified in Table E-5. Each column must have an identified model in order to complete the required modeling steps to quantify changes in that endpoint. In cases where defensible links are not quantified, opportunities exist for qualitative analysis.

Table E-7 summarizes the quantitative and qualitative analyses that we propose based on the available model coverage. Geographic scope plays an important role in determining the level of analysis, such as a national assessment, a case study or a qualitative description that is possible given existing models. This exhibit demonstrates that, of the great number of known impacts of air pollution, only a subset can be assessed. In the next section we discuss the methods, results, and caveats of the analyses of these selected endpoints.

Table E-7
Summary of Endpoints Selected for Quantitative Analysis

Endpoint	Analysis	Geographic Scope
Lake acidification impacts on recreational fisheries	Quantification of improved fisheries with monetization of recreational value	Case study of New York State
Estuarine eutrophication impacts on recreational and commercial fisheries	Quantification of improved fisheries with monetization of avoided costs of alternative eutrophication control methods	Illustrative calculations for case studies of Chesapeake Bay, Long Island Sound, and Tampa Bay (with extensions to East Coast estuaries)
Ozone impacts on commercial timber sales	Quantification of improved timber growth with monetization of commercial timber revenues	National assessment
Ozone impacts on carbon sequestration in commercial timber	Quantification of improved carbon sequestration	National assessment
Toxicity impacts on recreational fishing	Qualitative analysis of improved recreational fisheries	Qualitative regional case studies of New York and Tennessee

Extending Future Analyses

By focusing on the readily measured impacts identified in Table E-7, it is possible to lose sight of ecosystem-level changes that may threaten ecosystem integrity in ways that alter or increase the risk of changing ecosystem structure and function. The isolation of service flows may often imply an oversimplified cause and effect relationship between pollution and the provision of the service flow, when more often the service flow is affected by complex non-linear relationships that govern ecosystem structure and function. Economic analyses that focus on a narrow class of acute service-flow impacts will not cover larger ecosystem-wide impacts that may ultimately prove most relevant to environmental policy decision making. This analytical weakness becomes apparent when impacts to ecological functions such as nutrient cycling and biological diversity are assessed.

Issues on which to focus future analytic work in this field include:

- Major linkages of cause and effect between air pollution and subtle deterioration in ecosystem integrity are difficult to quantify;

- Degradation of ecosystem integrity most often does not cause immediate measurable declines in ecosystem service flows that are monetarily valued by society;

- The time-frame required for many ecological impacts to manifest themselves is such that the present value of these impacts discounts to negligible sums; and,

- Uncertainties associated with the scale of complex ecological impacts are too great to allow for reliable estimation of the economic implications.

Because of the weaknesses in the available methods and data, the benefits assessment in this appendix does not represent a comprehensive estimate of the economic benefits of the CAAA. Moreover, the potential magnitude of long-term economic impacts of ecological damages mitigated by the CAAA suggests great care must be taken to consider those ecosystem impacts that are not quantified here. Significant future analytical work must be performed to build a sufficient base of knowledge and data to allow the expansion of this benefits assessment.

Eutrophication of Estuaries

This analysis considers the economic benefits of reduced nitrogen deposition and the effects on selected eastern estuaries attributable to the 1990 Clean Air Act Amendments (CAAA). Note that these estimates were not included in the primary benefits of the CAAA; these are presented here as an alternative calculation only. We present a description of how nitrogen deposition affects estuarine ecosystems, an explanation of the effects on ecological service flows, and an assessment of the benefits of reducing nitrogen deposition in the context of several case studies using avoided-damage and displaced-cost approaches as alternative estimates of benefits. A more comprehensive description of this analysis is found in *Benefits Assessment of Decreased Nitrogen Deposition to Estuaries in the United States Attributable to the 1990 Clean Air Act Amendments, 1990-2010* (IEc, 1999a).

Impacts of Nitrogen Deposition on Estuaries

Atmospherically derived nitrogen makes up a sizable fraction of total nitrogen inputs in estuaries in the eastern United States. When atmospheric nitrogen enters estuaries it can cause *eutrophication*, or an increased nutrient load that, in excess, changes the ecosystem's structure and function and affects the provision of ecological service flows. The ecological effects and their associated service flows are listed in Table E-8.

Table E-8
Service Flows Affected by Changes in Estuarine Ecosystems

Ecosystem Changes	Service Flow Impacts
Deterioration of breeding grounds for fisheries	Commercial fishing yields, species mix Recreational fishing catch rate, species mix
Loss of habitat for aquatic and avian biota	Existence value of a healthy estuarine ecosystem Wildlife viewing Aesthetics

Derivation of dose-response relationships between atmospheric nitrogen loading and ecological effects is complicated by the dynamic nature of ecological systems. In addition to being characterized by non-linear, "threshold" type responses, esturine ecosystems are simultaneously influenced by a variety of stressors (both anthropogenic and non-anthropogenic). This makes it difficult to quantify the nature and magnitude of ecological changes expected to result from a change in a single stressor such as nutrient loading. Further, if the state of the ecosystem has changed (as from oligotrophic[6] to eutrophic) the removal of the initial stressor does not necessarily mean a rapid return to the prior state. This complicates the quantitative benefits assessment of controlling nitrogen deposition through the CAAA.

Economic Benefits of Decreasing Atmospheric Deposition of Nitrogen

EPA's analysis begins with a geographic information system (GIS) approach to estimate the total volume of nitrogen inputs that the CAAA would reduce to three major estuaries, the Chesapeake Bay, Long Island Sound, and Tampa Bay. Unfortunately, resource limitations prevented us from examining more than three estuaries at this point. The three estuaries were chosen for several reasons. First, each of these areas maintains an active research center, either under Clean-Water Act or National Estuary Program provisions, and has identified airborne nitrogen as a major factor in its efforts to limit eutrophication. Second, each has conducted some research into the level of damages associated with nitrogen eutrophication, although in the final analysis only the Chesapeake Bay Program's research was sufficient to establish some measure of avoided damages. Third, each of these estuaries has in place a binding commitment to meet its nitrogen reduction targets, a necessary condition for applying the avoided costs approach. In these estuaries, failure to reduce airborne nitrogen deposition (as would be the case if no CAAA were in place) would imply that additional nitrogen reductions would be necessary from other sources, such as point or nonpoint surface water discharges. Implementation of the CAAA therefore effectively avoids the imposition of costs to achieve those nitrogen discharge reductions.

For each of the three estuaries selected, we then conduct two types of analyses. First, we assess the avoided nitrogen deposition loadings to the watershed and avoided costs from reducing nitrogen deposition, using submerged aquatic vegetation as a key biophysical indicator. Second, we estimate the avoided cost of implementing planned alternatives to the CAAA for reducing nitrogen deposition.

GIS-Based Deposition and Loadings Estimates

The first step toward calculating deposition-related nitrogen loadings to the three estuaries is to estimate the total deposition of nitrogen to the watersheds. Table E-9 presents our estimates of the quantity of nitrogen deposited to the Chesapeake Bay, Long Island Sound, and Tampa Bay

[6] Oligotrophy refers to a state of relatively low nutrient enrichment and low productivity of aquatic ecosystems. In contrast, eutrophy refers to a state of relatively high nutrient loading and higher productivity, sometimes leading to overenrichment and reduction in ecological service flows via water quality decline.

Table E-9
Total Nitrogen Deposition Based on GIS Analysis
(millions of lbs.)

Watershed	Scenario 1: 1990	Scenario 2: 2010 without CAAA	Scenario 3: 2010 with CAAA	Difference
Chesapeake Bay	345.1	452.4	258.1	**194.3**
Long Island Sound	78.3	93.7	56.8	**36.9**
Tampa Bay	8.1	11.3	7.0	**4.3**

watersheds. We present data for three different scenarios. The first scenario is our estimate of the quantity of nitrogen deposited in 1990, prior to the introduction of the CAAA. Scenario 2 is our estimate of the quantity of nitrogen deposited in 2010 without the CAAA, and Scenario 3 is the quantity deposited with the CAAA. The difference between Scenarios 2 and 3 represents the potential future impacts of the CAAA on nitrogen deposition.[7]

As the exhibit indicates, the CAAA are likely to have a significant impact on the quantity of nitrogen deposited to each of the three watersheds. For the Chesapeake Bay watershed, nitrogen deposition is expected to be nearly 195 million pounds less in 2010 (43 percent) than it would have been without the CAAA. For the Long Island Sound and Tampa Bay watersheds, this figure is approximately 37 million pounds (39 percent) and four million pounds (38 percent), respectively.

We also estimate the prevalence of major categories of land use in each of the three watersheds. Land use is a critical component of our analysis because the quantity of nitrogen runoff that eventually reaches the estuary varies according to the type of land that receives the atmospheric deposition. For example, the fate of atmosphericly deposited nitrogen will differ if the nitrogen falls on forest versus urban land, because forest land generally retains a greater percentage of nitrogen than urban land. Our analysis uses distinct nitrogen "pass-through" figures for each category of land use. Pass-through represents the percentage of atmospherically deposited nitrogen that is ultimately transported to surface water rather than retained by the land.

Table E-10 presents the land use prevalence and the pass-through factors that we use for each of the three watersheds in our analysis.[8] As the exhibit indicates, forests (53 percent) and agricultural lands (32 percent) represent the majority of the land use in the Chesapeake Bay watershed. In the Long Island Sound watershed, forests (67 percent) again dominate land use; however, urban lands account for as much territory as agricultural lands (11 percent). For the Tampa Bay watershed, agricultural lands constitute the largest land use (33 percent), while rangelands (19 percent), urban land (15 percent), and wetlands (10 percent) represent a much greater proportion of the land use than in the other two watersheds.

[7]These data are derived from IEc's spatial analysis of the watersheds and RADM nitrogen deposition modeling. The RADM modeling is described in Appendix C of this report.

[8] Pass-through estimates were derived from EPA's analysis of the relevant literature - see IEc (1999).

Table E-10
Land Use Prevalence and Pass-Through Figures

Watershed	Forest	Agricultural	Urban	Water	Wetlands	Other*
Chesapeake Bay						
Land Use	53%	32%	6%	7%	1%	1%
N Pass-Through	20%	30%	50%	100%	20%	30%
Long Island Sound						
Land Use	67%	11%	11%	9%	2%	0%
N Pass-Through	20%	30%	50%	100%	20%	30%
Tampa Bay						
Land Use	5%	33%	15%	15%	10%	22%
N Pass-Through	20%	30%	50%	100%	20%	30%

* "Other" areas in Tampa Bay include rangeland (19%) and barren land (3%).

We use the pass-through figures and land use prevalence in conjunction with deposition quantities to estimate nitrogen loadings to each estuary. Table E-11 displays our nitrogen loadings estimates for the three watersheds under the three scenarios. As the exhibit indicates, loadings from atmospheric deposition decrease significantly due to the CAAA. For Chesapeake Bay, for example, we estimate that nitrogen loadings with the CAAA will be approximately 79 million pounds in 2010, approximately 58 million pounds less than our estimate for loadings in 2010 without implementation of the CAAA. For the Long Island Sound and Tampa Bay, the difference between the two scenarios is approximately 13 million pounds and 1.8 million pounds, respectively.

Table E-11
Nitrogen Loadings from Atmospheric Deposition
(millions of lbs.)

Watershed	Scenario 1: 1990	Scenario 2: 2010 without CAAA	Scenario 3: 2010 W/CAAA	Difference
Chesapeake Bay	105.2	137.5	79.4	58.1
Long Island Sound	26.7	31.9	19.1	12.8
Tampa Bay	3.4	4.7	2.9	1.8

Displaced Costs from Reducing Atmospheric Deposition to Estuaries

It is possible to use a displaced cost approach to determine the benefits associated with reduced nitrogen emissions. To reduce excess nutrient loads (including nitrogen) to local estuaries, many coastal communities are pursuing costly abatement options. These options include point source controls as well as urban non-point and agricultural non-point source controls. We estimate the marginal costs of abatement associated with these controls as implemented in the three case study estuaries. To the extent that nitrogen deposition can be controlled more cost effectively than point-source discharges, the control expenditures displaced by the CAAA represent a benefit to society.

Ideally, a nitrogen management program would result in the least expensive abatement possible, thereby minimizing the resources society expends on nitrogen control. The lowest marginal cost pollution reduction is exploited first, and pursued to its limit before the next least costly alternative is exploited, and so on until the required nitrogen reduction is met (represented by the dark columns in Figure E-1).

With the CAAA, a portion of the resources society committed to or would have committed to reducing a quantity of waterborne nitrogen may be unnecessary. Following the cost minimizing strategy, society will forego the most expensive control cost option first, pursue it to its limit before the next most expensive alternative is foregone, and so on until the nitrogen reduction benefits from the CAAA are exhausted (represented by the lightly shaded columns in Figure E-2). The level of nitrogen reduction remaining will therefore be accomplished at the lowest cost (represented by the dark columns in Figure E-2).

Figure E-1. Nitrogen Reduction Without CAAA

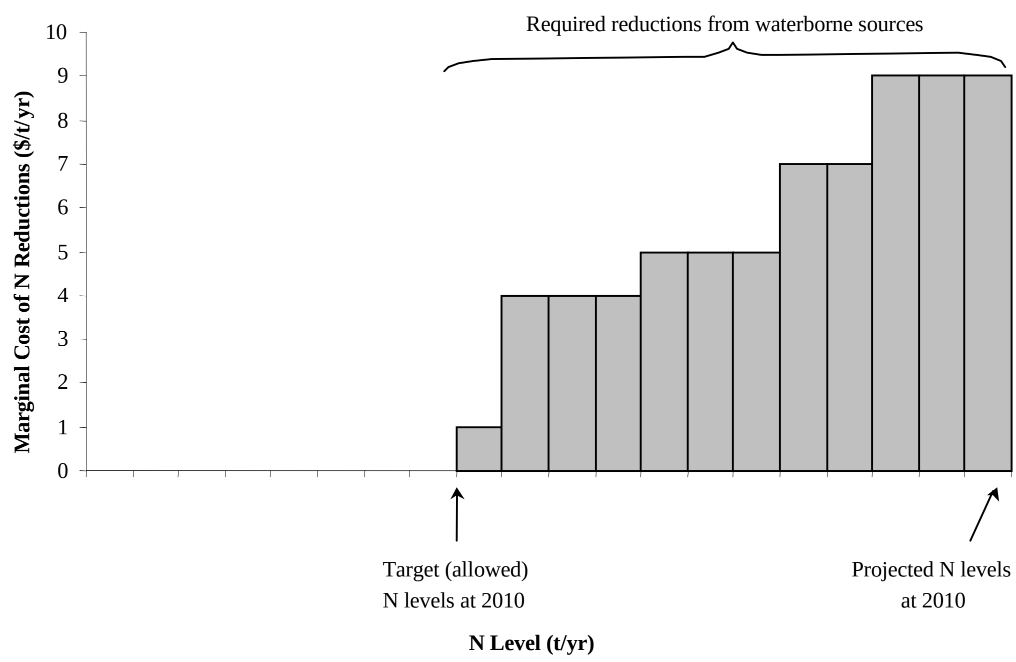
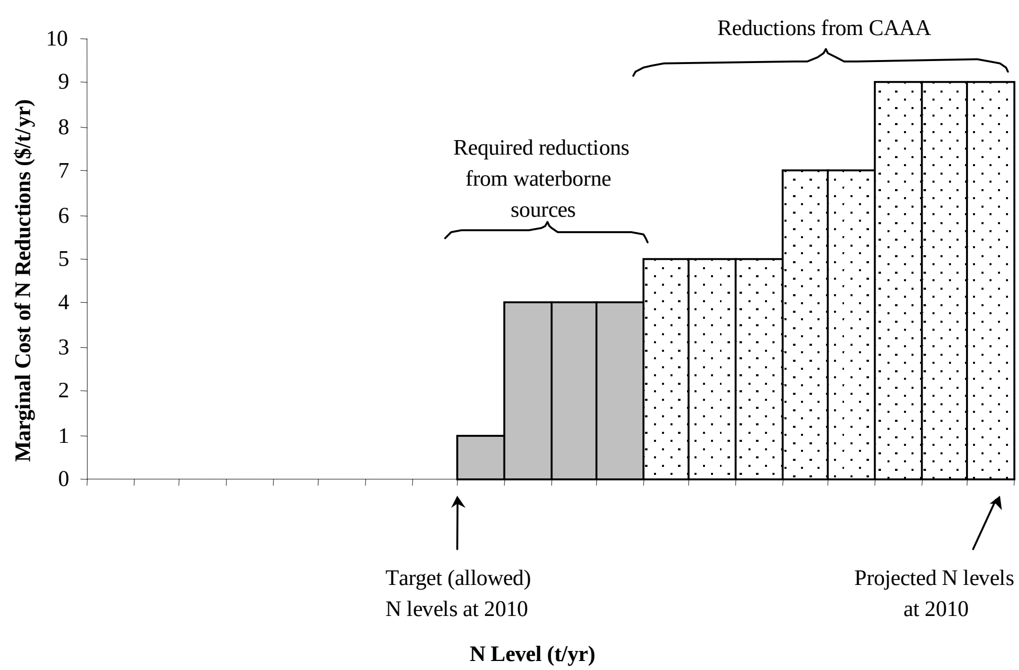


Figure E-2. Nitrogen Reduction With CAAA



We develop our avoided-cost estimate by assuming that decision makers will choose to forego the most costly nitrogen abatement projects first. That is, we assume that reduced deposition and the resulting loadings reduction will eliminate the need for additional point or non-point source controls at the high end of the marginal cost curve.

To estimate the economic benefits of reduced nitrogen deposition from the CAAA, we require site specific information from the watershed level. A justifiable avoided cost analysis relies upon the existence of realistic and enforceable nitrogen reduction goals for each estuary. Without specific targets or reduction goals, it is not possible to suggest that there are any control costs to be avoided. As described earlier, we have chosen case study estuaries that fit this criterion. These areas have established nitrogen reduction programs that rely primarily on reductions of effluent from point sources as well as reductions in non-point source discharges. Information on the reduction goal and potential abatement options for meeting those goals allow us to estimate the portion of the goal that can be met by the CAAA, as well as the associated cost savings.[9] We summarize those results in Table E-12.

Next, we need to know the annual quantity of atmospheric nitrogen deposited on the watershed. Last, we need to understand details about the different nitrogen reduction programs that could be implemented in the watershed. This includes the quantity of nitrogen reduced through a particular control option (e.g., agriculture best management plans[BMPs]), and the unit cost of reducing that nitrogen (i.e. dollars per pound or ton of nitrogen reduced).

The benefits valuation derived using the avoid-costs approach should be interpreted cautiously for two reasons. First, it is an estimation of capital costs that serve more purposes than mitigating nitrogen inputs into the estuaries of concern. Water treatment works are intended to provide waste water treatment for a variety of pollutants and may be required even in the absence of air deposition of nitrogen. Second, the nitrogen loading targets for the estuaries are not concrete, strictly enforced limits, based on certain knowledge of the capacity of the estuaries to accept nitrogen inputs. Instead, the targets may change over time as knowledge of the effects of nitrogen to these estuaries change. For these reasons, we do not include these estimates in the primary benefits estimates for the CAAA.

[9]With increasing populations, controls of alternative sources (e.g., automobile and utility emissions) may be needed simply to meet the original target or goal; if the CAAA amendments are necessary just to achieve the target reductions, then we are actually measuring alternative costs and not displaced costs.

Table E-12
Estimated Avoided Costs For Three Estuaries

Estuary	Reduced N Deposition in 2010(millions of pounds)	Lower-BoundMarginal Cost($/lb/yr.)	Upper-Bound Marginal Cost ($/lb/yr.)	Estimated Annual Avoided Costs in 2010 ($millions)
Long Island Sound	12.8	$2	$8	$25.6-$102.4
Chesapeake Bay	58.1	$6	$22	$349-$1,278
Tampa Bay	1.8	$6	$38	$11 - $68

Results for Case Study Estuaries

- Under the *Chesapeake Bay* Agreement, the signatories (EPA, Maryland, Virginia, Pennsylvania, and the District of Columbia) have agreed to reduce nutrient loadings to the Bay by 40 percent by the year 2000, relative to 1985 levels. This goal translates to a nitrogen reduction of about 186 million pounds per year. A great deal of progress toward this goal has already been made, although success differs across sub-regions in the watershed (Chesapeake Bay Program 1997). Nitrogen loadings reductions achieved thus far are the result of both point and non-point source controls. Since 1985, 33 of the 315 major municipal treatment plants in the region have upgraded to biological nutrient removal (BNR), an advanced treatment technology specifically focused on nitrogen removal. These upgrades have reduced annual loadings by about 13 million pounds (as of 1996). Approximately 60 additional facilities are expected to implement BNR in the future. In addition, agricultural and urban best management practices have reduced non-point source loadings by about 16 million pounds per year, with additional implementation of BMPs planned for coming years (Chesapeake Bay Program, 1997). Because both point and non-point source controls play a role in anticipated future nitrogen reductions in Chesapeake Bay, we develop marginal and avoided cost estimates that incorporate both types of control. As reflected in Exhibit 15, we estimate avoided cost benefits for Chesapeake Bay ranging from about $349 million to $1.3 billion.

- *Long Island Sound* has established a goal of reducing nitrogen loadings by approximately 48 million pounds by 2015. Point source controls are anticipated to be the primary source of these reductions. Numerous sewage treatment plant upgrades are slated for the region, many of which are currently under construction. We use data from the Connecticut Department of Environmental Protection and New York Department of Environmental Conservation. The marginal cost figures yield an estimate of avoided costs that ranges from about $26 to $102 million per year.

- In 1996, the *Tampa Bay* Estuary Program (TBEP) adopted a five-year nutrient management goal that caps annual nitrogen loadings at 1992-1994 levels. Nitrogen loading to Tampa Bay is expected to increase seven percent by the year 2010 as a result of population growth and related commercial and residential development. To offset this growth and maintain current nitrogen levels, the TBEP has asked local governments, agencies, and industries to reduce total nitrogen loadings to the Bay by approximately 84 tons (168,000 pounds) per year by the year 2000. The result of this planning effort is the Nitrogen Management Action Plan (TBEP, 1998). This plan lists the projects undertaken or planned by industry, local governments, and agencies that will contribute to meeting the nitrogen management goal for 2000 and beyond. These projects, which together surpass the nitrogen reduction goal for Tampa Bay, are a combination of point and non-point source control measures. The non-point source control projects include urban stormwater retention ponds, wetlands restoration, and land acquisition. The point source projects focus on advanced treatment technologies such as BNR at (publically owned treatment works (POTWs). For example, one project, proposed for implementation after the year 2000, will involve additional treatment of effluent from a POTW prior to reuse in the regional water supply. We estimate annual avoided costs for Tampa Bay ranging from about $11 million to $68 million.

These three estuaries represent only a portion of the total estuarine area affected by nitrogen deposition in the United States. The Chesapeake Bay and Long Island Sound account for roughly 20 to 25 percent of the East Coast estuarine watershed area addressed by the National Estuary Program, and Tampa Bay is a small fraction of the total Gulf Coast estuarine watershed area. As a result, our estimates reflect only a partial analysis of the national impact of nitrogen deposition.

Results for Total East Coast Estuarine Area

To extrapolate the results of this analysis to all East Coast estuaries, we assume all estuaries along the Atlantic Coast have binding nitrogen budgets. We then use the same geographic information system (GIS) approach we used in our analysis of Chesapeake Bay, Long Island Sound, and Tampa Bay to estimate total nitrogen deposition and the associated loadings to estuaries located along the Atlantic Coast. This approach allows us to estimate nitrogen loadings in the year 2010 with and without CAAA emissions controls. Since watershed specific nitrogen control program information is not available for each watershed, we extrapolate BMP cost and nitrogen reduction data from the three case studies across all East Coast watersheds.[10]

Although we simplistically assume that each estuary has a nitrogen budget, the total East Coast displaced cost analysis does not include all estuaries along the Atlantic Coast, since some estuaries are not sensitive to nitrogen loadings. Certain estuaries are able to process large amounts of nutrients (nitrogen and phosphorus) without problems, while others are unable to process even low amounts of nutrients. Rather than rely on nitrogen levels to determine which estuaries to include in the displaced cost analysis, we use a measure of eutrophic susceptibility developed by Bricker et al. (1999-DRAFT), to determine how sensitive estuaries are to nitrogen loadings. Using their eutrophic susceptibility classification (low, medium, and high) we then exclude estuaries with low eutrophic susceptibility from the displaced cost analysis.[11]

The total displaced costs for the East Coast is simply the sum of the displaced costs for each estuary along the Atlantic Coast classified as moderately or highly susceptible to eutrophication (see Table E-13).[12] The lower-bound estimate of $261.5 million represents a point source control strategy. The upper-bound estimate of $2,766 million represents a strategy

[10] The control cost and nitrogen reduction data is available in the IEc memorandum to EPA dated August 26, 1999.

[11] This index is based on a variety of factors influencing sensitivity of estuaries to eutrophication, including water surface area of the estuary, estuary volume, freshwater inflow, tidal cycles, and vertical stratification.

[12] In the North Atlantic Region, Blue Hill Bay, Casco Bay, Englishman Bay, Kennebec/Androscoggin River, Merrimack River, Muscongus Bay, Narraguagus Bay, Penobscot Bay, and Saco Bay have a low susceptibility to eutrophication and are excluded from the displaced cost analysis. All estuaries in the Mid Atlantic Region are moderately to highly sensitive to eutrophication. In the South Atlantic Region, New River, Ossabaw Sound, Savannah River, and St. Helena Sound have a low susceptibility to eutrophication and are not included in the displaced cost analysis.

Table E-13
Avoided Cost for Atlantic Coast

Watershed	Reduction through CAAA, Millions of Pounds (2010)	Avoided Cost, 100 Percent from Point Source ($1000/yr.)	Avoided Cost, from Nonpoint Source First, then Point Source on Difference ($1000/yr.)
North Atlantic			
Cape Cod Bay	0.51	1,282	7,915
Great Bay	0.57	1,446	19,130
Massachusetts Bay	1.19	3,015	66,887
Sheepscot Bay	0.13	340	2,397
Sub-Total	13.02	6,083	96,329
Mid Atlantic			
Barnegat Bay	0.48	1,202	19,688
Buzzards Bay	0.78	1,976	14,429
Chincoteaque Bay	0.33	824	3,690
Delaware Bay	12.11	30,640	384,002
Delaware Inland Bays[1]	0.18	459	4,592
Gardiners Bay	0.64	1,624	13,929
Great South Bay	1.30	3,294	78,682
Hudson River/Raritan Bay	11.15	28,197	460,659
Narragansett Bay	1.65	4,176	65,841
New Jersey Inland Bays	1.12	2,825	28,314
Sub-Total[2]	29.74	75,219	1,073,826
South Atlantic			
Albemarle Sound	16.29	41,223	272,772
Altamaha River	7.24	18,315	182,591
Biscayne Bay[3]	0.78	1,973	12,136
Bogue Sound	0.30	756	9,426
Broad River[3]	0.46	1,160	7,137
Cape Fear River	6.59	16,674	171,805
Charleston Harbor	10.58	26,764	306,698
Indian River	0.76	1,926	36,045
North/South Santee Rivers	0.32	821	3,671
Pamlico Sounds	9.58	24,225	191,289
St. Andrew/St. Simons Sounds[3]	1.18	2,974	18,297
St. Catherines/Sapelo Sounds	0.28	721	6,525
St. Johns River	4.72	11,940	152,909
St. Marys River/Cumberland Sound	0.53	1,338	8,886
Winyah Bay[1]	11.63	29,415	215,612
Sub-Total	81.07	180,225	1,595,799
Total East Coast	**123.82**	**261,527**	**2,765,954**

[1] CAAA N reductions met with agriculture, forestry, and urban BMPs (point source reductions not required).
[2] Excluding Long Island Sound and Chesapeake Bay.
[3] CAAA N reductions met with agriculture BMPs (further reductions not required).

of nonpoint source BMPs, with further nitrogen reductions from point sources, if required.[13]

This estimate is based on an extrapolation of nitrogen abatement costs from representative watersheds. Although this is a broad assumption, it does provide a gross estimation of the range and magnitude of the CAAA benefits for the Atlantic Coast as a whole and its component watersheds.
Due to our general assumptions, a high degree of uncertainty is associated with this range. First, we do not know enough about the nature of the nitrogen budgets for each estuary and if those budgets would be binding. If nitrogen budgets are not binding, these regions may have little incentive to reduce nitrogen loadings. Furthermore, because of the lack of watershed specific cost information, we rely on available abatement cost data from the Chesapeake Bay and Long Island Sound to represent point source and nonpoint source unit abatement costs for all watersheds along the Atlantic Coast. The tightness of the range and accuracy of the displaced cost analysis is dependant on an accurate understanding and representation of the nitrogen abatement costs associated with the different point source control options and nonpoint source BMPs in each individual watershed. Lastly, we use marginal cost figures that represent averages of the costs associated with certain control measures. For example, our agriculture BMP cost figure is a simple average of six different agriculture BMPs, each with a different level of nitrogen reduction and a different cost per pound of nitrogen reduced. While beyond the scope of this effort, a more refined analysis would use marginal cost estimates based on the precise mix of agriculture BMPs to be implemented in each watershed and the cost of each, as determined by local factors within the watershed.

[13] In our analysis of the total displaced costs for the Atlantic Coast, several watersheds meet their CAAA nitrogen reduction levels without relying on point source controls.

Avoided Damages to Estuarine Ecosystems

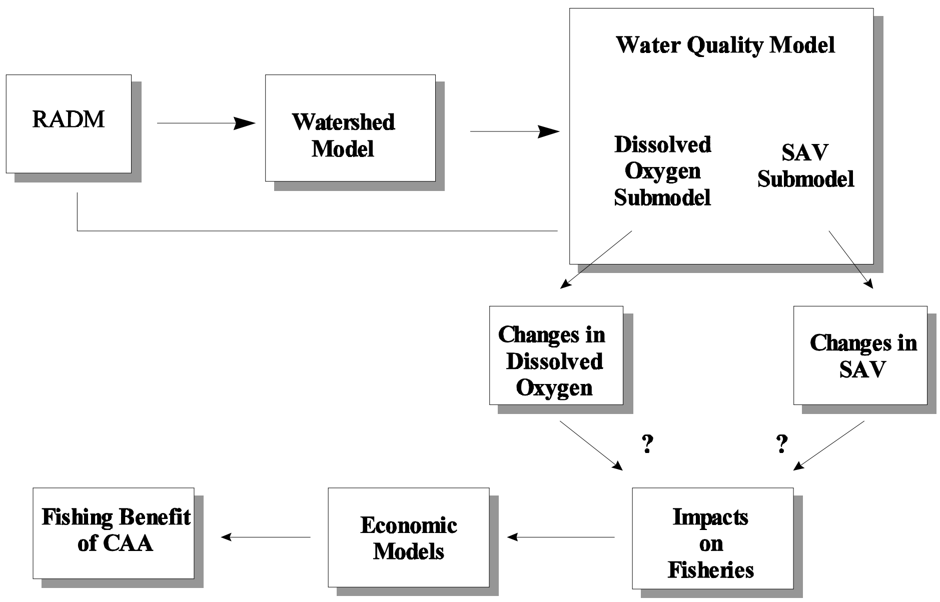
Theoretically, a modeling system that describes water quality changes, including fish population dynamics as a function of nitrogen input, would provide an assessment of the avoided damages from mitigating nitrogen deposition (Figure E-3). Because of current modeling and data constraints, however, the only means to quantify the damages of eutrophication from nitrogen deposition is through the use of specific biophysical indicators of estuarine health. Changes in an indicator, such as aerial extent of seagrass beds, can measure habitat damage. Based on changes in habitat, the change in ecosystem service flows associated with that habitat can be estimated.

From an economic perspective, this approach is useful in cases where habitat is closely related to the provision of ecological service flows, such as commercial and recreational fishing yields. Using seagrass beds as an indicator, we describe the potential for avoiding estuarine damages through the CAAA.

Submerged aquatic vegetation (SAV) is a group of angiosperms (plants that bear seeds, as opposed to algae, which reproduce via cell division) that form extensive meadows, providing habitat, breeding grounds, and nursery for a variety of organisms including fish, birds, shellfish, and invertebrates (Jacobs et al. 1981; Bell et al. 1989; Howard et al. 1989; Burkholder et al. 1992; Orth et al. 1994). SAV meadows give considerable three-dimensional structure to the seabed that provides small organisms with a place to hide from predators, acts as a sediment trap, and functions as a breakwater offering natural shoreline protection (Vermaat et al. 1998). Along with aerial extent, the density of SAV may be important in defining the health of the SAV community.

Though a universal nitrogen-SAV relationship has not been derived, field data show that increased nitrogen loading has been accompanied by extensive decline in SAV in a variety of estuaries (Valiela et al. 1997; Burkholder et al., 1992; Coastlines, 1994; Vermaat, 1998). In the Chesapeake, SAV acreage declined from more than 76,000 to about 40,000 acres

Figure E-3
Estuary Models and Ecological Impacts of Concern



between 1870 and 1950, concurrent with increased nutrient loading (Coastlines, 1994). From 1950 to 1980, when nutrient loading took a sharp upward turn, the decline continued to a low point of about 21,000 acres. This decline is quite likely due to shading by excessive algal growth and increased growth of epiphytic plants that also respond positively to increased nitrogen availability (Coastlines, 1994).

Empirical evidence from Tampa Bay and the Chesapeake Bay show that SAV populations will likely benefit from reduced nitrogen inputs to these estuaries. In Tampa Bay, there is a notable correlation between SAV and nitrogen inputs, as described in Figure E-4.[14]

In Chesapeake Bay, it is not possible to identify a statistically significant relationship between these two variables in isolation because of the large and heterogenous nature of the Chesapeake Bay. Embayments vary greatly in physical characteristics including depth and salinity, complicating the relationship. Nonetheless, the general trend has been an increase in seagrass acreage and a decrease in nitrogen loading. Figure E-15 shows the change in SAV acreage over the past two decades.

Because the relationships we describe from existing data are not sufficiently robust to allow projections of SAV coverage as a function of nitrogen deposition in our scenarios, we utilize alternative methods to provide quantitative benefits estimates in the following section.

[14] Two factor analysis of variance (ANOVA) analysis confirms this correlation; see EPA 1999 for more details.

Figure E-4

Nitrogen Load vs. Seagrass Acreage in Tampa Bay

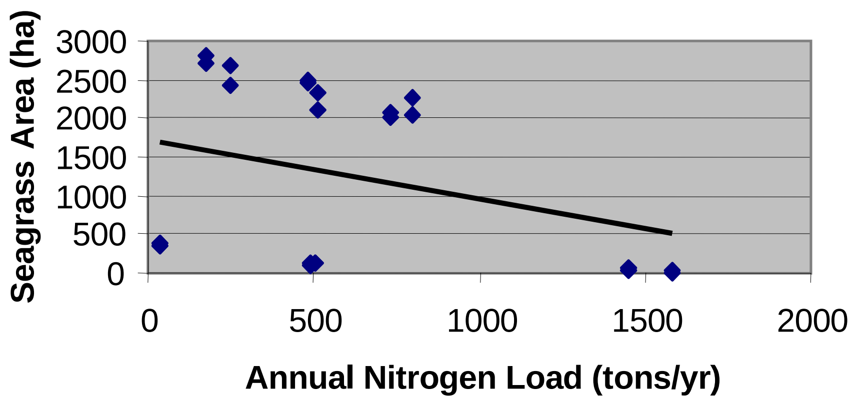
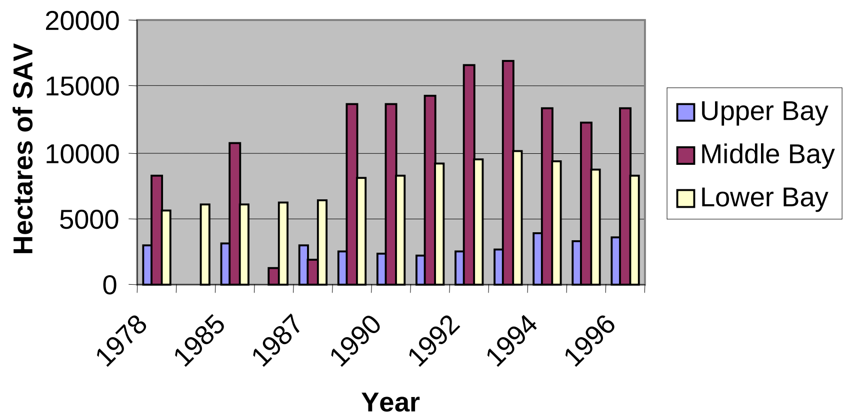


Figure E-5

Chesapeake Bay SAV
1978-1996



Caveats and Uncertainties

Though it is difficult to directly predict the nature and magnitude of the ecosystem impacts of eutrophication, we conclude that continued nitrogen inputs at current levels will result in deleterious effects. The major caveats and uncertainties associated with these analyses follow.

- Our nitrogen loading estimates are derived using a highly simplified approach that takes into account total deposition and the nitrogen retention characteristics of different land uses. Some factors suggest that we may overstate loadings because we do not consider the effects of nitrogen travel through varied distances and heterogeneous geography, such as rivers and streams. For example, an additional 20 to 75 percent of nitrogen is retained during transport in rivers and streams (Hinga, et al., 1991).

  Other factors suggest that we may understate loadings. Most significantly, the Geographic Information Retrieval Analysis System (GIRAS) land use data in our GIS analysis were compiled in the early 1980s. It is likely that the current amount of urbanized land is greater than these data indicate. Furthermore, continued development of forest and other open land suggests that land uses will change significantly in the period between now and 2010. Because nitrogen removal in urban land is low, more refined land use data for future years would likely lead to greater estimates of nitrogen loadings. We compare our nitrogen loading estimates with those of estuary programs and published literature for 1990 to verify whether our approach generates results that correspond with existing estimates. For the Chesapeake our estimate of 150 million tons falls in the middle of existing estimates ranging from 58 to 159 million tons (EPA 1997e; Patwardhan and Donigian 1997; Fisher et al. 1998; Tyler 1998) for Long Island Sound our estimate of 27 million tons corresponds with the existing estimate of 26 million tons (Stacey 1998), and for Tampa Bay our estimate of 3 millions tons is close to the existing estimate of 2 million tons (TBEP 1998; Zarbuck et al. 1994).

- We base our estimates of avoided costs on simplified assumptions regarding the control measures that would be eliminated as a result of reducing atmospheric nitrogen. The mix of nitrogen controls that could be displaced will be influenced by state regulations affecting treatment plants and non-point sources as well as by pollution reduction goals for different sub-basins in each watershed. For example, water quality objectives for pollutants other than nitrogen may require controls that we assume could be eliminated.
- Similarly, we use marginal cost figures that represent averages of the costs associated with control measures. For example, we apply generic non-point source control cost estimates based on a mix of agriculture, forestry, and urban best management practices. While beyond the scope of this effort, a more refined analysis would use marginal cost estimates based on the precise mix of best management practices to be implemented in each watershed and the cost of each, as determined by local factors such as levels of nitrogen in soils, evolving agricultural practices, and changing development patterns.

Acidification of Freshwater Fisheries

During the 1970s and 1980s, "acid rain" came to be known to the public as a phenomenon that injures trees, forests, and water bodies throughout Europe and in some areas of the United States and Canada. One of the goals of the CAAA was to address the problem of acidification of terrestrial and aquatic ecosystems caused by acidic deposition. In this section we evaluate economic benefits accruing to society as a consequence of reductions in emissions of sulfur and nitrogen oxides mandated by the CAAA. In particular, we focus on a quantitative analysis of benefits derived from a reduction in acidification of aquatic bodies as they relate to recreational fishing in the Adirondacks region of New York State. Our analysis indicates that by mitigating acidification with the regulations promulgated under the CAAA, cumulative benefits between 1990 and 2010 can be accrued in the range of $67 to $465 million. Using the results from the acidification estimates, it is also plain to see that the CAAA may be preventing further ecological impacts from acidification that are not quantifiable in economic terms using available methods. A more comprehensive description of this analysis is found in *Economic Benefits Assessment of Decreased Acidification of Fresh Water Lakes and Streams in the United States Attributable to the 1990 Clean Air Act Amendments*, 1990-2010 (IEc, 1999b).

Acidification of Surface Waters and Ecological Impacts

Acidification of surface waters is frequently described using two measures. One measure of acidification is pH, which is based on the hydrogen ion (H+) concentration found in surface waters.[15] The pH of a water sample can range from 1 to 14 on a logarithmic scale, with pure water having a pH of seven. The term acidic usually refers to a pH below seven, indicating high concentrations of hydrogen ions (H+). Rain water that is unaffected by anthropogenic factors (natural rain) is weakly acidic (pH 5.0 - 6.0), due to the presence of natural weak acids. With addition of acids from human activities, however, the pH of rain can range from 3.5 to 5.0 (NAPAP, 1991, p.15). Most freshwater lakes and streams have a pH between 6.5 and 8, indicating that surface waters can be naturally acidic. Only a small percentage of aquatic ecosystems are naturally acidic with pHs below 6.5, and concerns about anthropogenic acidification focus on the effects that may occur with decreases in pH below pH 6.5 (EPA, 1995a p. 9).

The second commonly used measure of acidification is Acid Neutralizing Capacity (ANC), which describes a water body's ability to neutralize acids added to the water column. Surface waters with higher ANC are generally more resistant to acidification and empirically tend to have higher pH levels. ANC is measured in micro-equivalents per liter (μeq/L). Surface waters with an ANC of less than 200 μeq/L are considered to have a low capacity for neutralizing acids. Water bodies with an ANC of 50 μeq/L or less have a very low capacity for neutralizing acids, and water bodies with an ANC of 0 μeq/L or less have no ability to neutralize acids and are acidic. These water bodies have no ability to neutralize acids and tend to be the most sensitive for long-term pH depressions below 6.0, which can produce the most severe effects on aquatic life (EPA, 1995a p. 9, NAPAP 1991 p.15).

Acidic deposition can lead to two kinds of acidification processes, depending on the duration of acidifying events. First, chronic acidification describes a situation in which acidic deposition leads to long-term changes in soil and water characteristics, causing chronically toxic environmental effects. Second, episodic acidification is a phenomenon in which surface waters experience short-term (hours to weeks) decreases in pH, usually during extreme hydrological events such as storm discharge or snowmelt (EPA, 1995a, p.9, NAPAP, 1991 p.18). For acid-sensitive fish species in some lakes or streams, for example, episodic events can cause complete spawning or recruitment failures (EPA, 1995a p.10).

The most comprehensive survey of surface waters comes from the National Surface Waters Survey (NSWS), which was conducted as part of the National Acid Precipitation Program (NAPAP). Based on the results from the NSWS surveys of lakes throughout the United States, an estimated 4.2 percent (1,180) of the NSWS lakes were acidic, defined as having ANC less than 0 μeq/l with pH levels in the range of 5.0 to 5.5. Nearly all of these lakes were in eastern portions of the United States, located in six "high-interest

[15] The pH of a water sample is equal to the negative log of its hydrogen ion concentration: pH= -log[H+].

regions". The six areas identified are New England, the Adirondacks, the mid-Atlantic Coastal Plain, the mid-Atlantic Highlands, Florida, and the upper Midwest. The NSWS found that acidic deposition is the dominant source of acid anions in about 75 percent of the acidic lakes and 50 percent of the acidic streams in their sample. Figure E-6 depicts the geographic ranges of known acidification.

The effects of acidity on aquatic organisms are determined by a number of different water quality variables, the most important of which are pH, inorganic aluminum, and calcium. The combined effects of these variables can adversely affect the physiology of individual organisms as well as population-level parameters. The direct effects may be classified as involving either recruitment failure or reduced adult survival. The outcomes on these impacts are declining acid-sensitive fish populations and a consequent decline in species richness (SOS/T 13 p.13-126).

Modeling Acidification

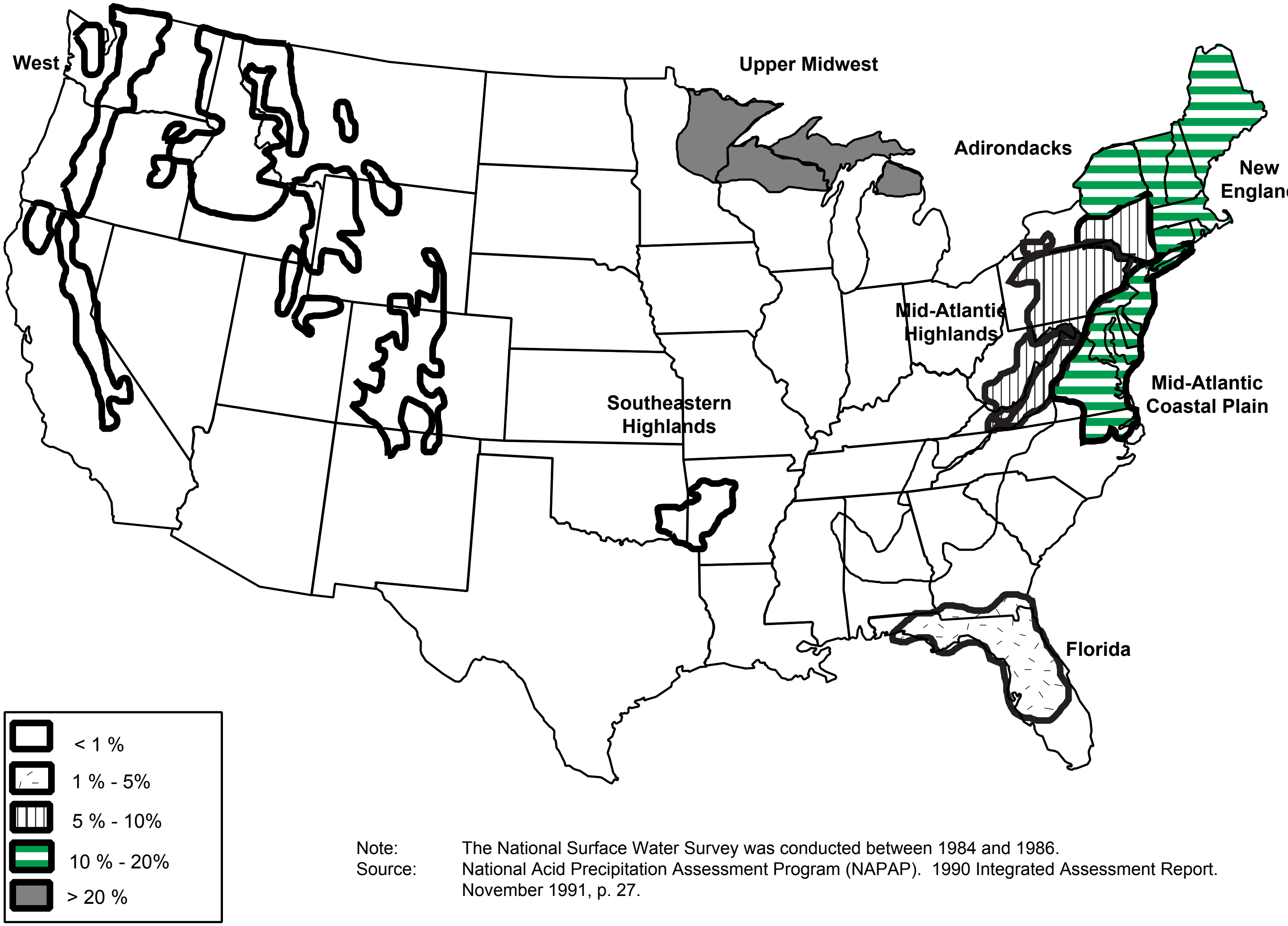
Figure E-7 shows the stages of modeling the ecological and economic impacts of acidification. This analysis uses RADM deposition data for the year 2010, and an extended deposition scenario that we develop for the year 2040, to demonstrate the possible lagged effects of the CAAA. We use these data in an acidification model that generates an estimate of the acidity of lakes in the Adirondacks. Lake acidity is input to an economic model that estimates the costs to anglers of diminishing lake water quality, and consequently declining fish populations.

We use the same emissions and deposition data for the period 1990 to 2010 as in each of our other endpoint analyses. The primary difference is that we extend the deposition scenarios to 2040 in order to accommodate for the lagged physical effects of acid deposition. This lag is a function of multiple watershed and water body characteristics influencing recovery from prolonged acidic deposition. Results of various efforts to model freshwater acidification showed that recovery of acidified water bodies can take over 50 years, even with substantial (up to 70 percent) reductions in sulfate deposition (Jenkins et al. 1990; Wright et al. 1994), while watershed soils may require 150-200 years for full recovery (Cosby et al. 1985).

Because appropriate data are lacking to simulate the emission and deposition of acidic pollutants between the years 2010 and 2040, we use two deposition scenarios for the period 2010-2040:

- Constant deposition from 2010 to 2040 at levels projected for 2010 under the regulations of the CAAA; and,
- Constant deposition from 2010 to 2040 at levels projected for 2010 without the regulations of the CAAA.

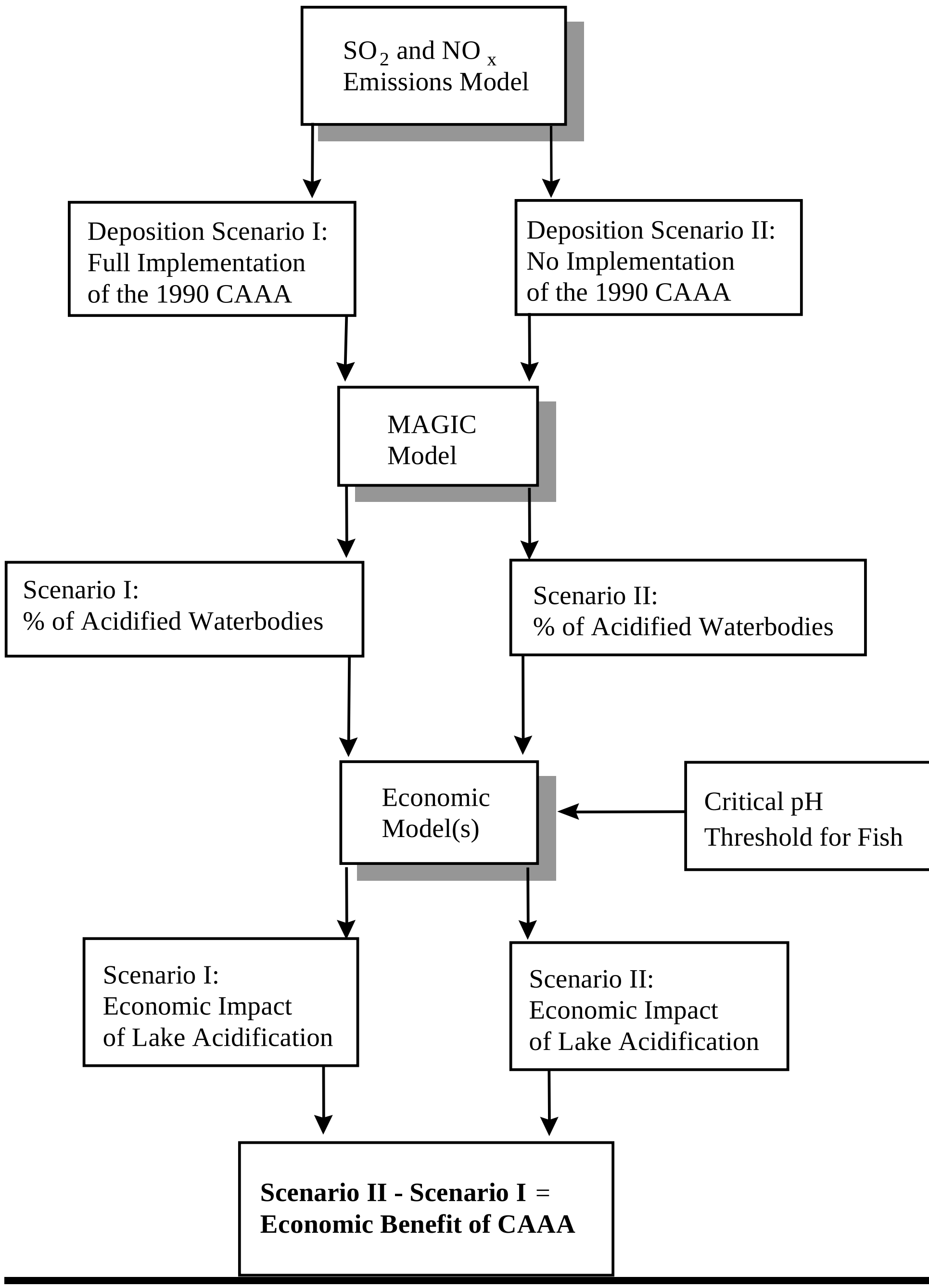
Figure E-6
Percentage of Acidic Surface Waters in the National Surface Water Survey Regions



Note: The National Surface Water Survey was conducted between 1984 and 1986.
Source: National Acid Precipitation Assessment Program (NAPAP). 1990 Integrated Assessment Report. November 1991, p. 27.

Figure E-7

Acidification of Freshwater Ecosystems



Using these deposition scenarios, we estimate the extent of acidification in the Adirondacks. To do this, we use the Model of Acidification of Groundwater in Catchments (MAGIC). MAGIC is a lumped parameter model that was originally developed to project the long-term effects (i.e., decades to centuries) to surface water caused by acidic deposition (Cosby et al, 1985a, 1985b). NAPAP and the Acid Rain Program of EPA use MAGIC extensively in analysis of acidification in the Eastern United States (Church et al, 1989; Church et al, 1992; EPA, 1995), the results of which have been rigorously peer-reviewed and used in previous policy analyses. The data that MAGIC produces describe the extent of acidification (i.e., pH and ANC levels) that will occur in a sample of sensitive lakes in the Adirondacks as a function of acidic deposition levels.

As mentioned earlier, capacities of watersheds to retain deposited sulfur or nitrogen containing compounds are among the most important factors influencing surface water acidification. Because increasing stages of nitrogen saturation are likely to lead to decreasing nitrogen-retention capacities, it is necessary to consider these effects in our modeling approach. MAGIC, however, currently does not explicitly represent detailed cycling or processes affecting the rate of nitrogen uptake and release because processes that control the transition of a watershed to a state of nitrogen saturation leading to surface water acidification are not well understood (Van Sickle and Church, 1995). Instead, we use a sensitivity analysis of two boundary conditions, representing a nitrogen-saturated watershed and a watershed where terrestrial ecosystems continue to utilize the majority of deposited nitrogen.

Our means to assess the economic impact of acidification is to measure the change in social welfare that results from reducing the quality of available lakes for recreational fishing. In order to accomplish this, we must determine the water chemistry parameters at which declines in recreational fish populations are perceptible by anglers. As mentioned earlier in this memorandum, the toxicity of low levels of pH to fish species depends on a variety of factors, including the concentration of inorganic aluminum and calcium in the water column as well as specific sensitivities of locally occurring fish species. The level of pH is not a precise indicator of the habitability of a lake for fish. But because the full complement of relevant water chemistry variables for lakes in the region is not available, we are forced to work with pH as a proxy for habitability. In order to accommodate for the variable effects of pH due to other water chemistry variables, we derive a range of pH values where negative effects on fish species are empirically known to occur. We summarize these results in Table E-14.

Table E-14
Summary of pH-Based Effects Threshold

	pH Effects Threshold (Low End)	pH Effects Threshold (High End)
Range for All Fish Species	4.2	5.8
Range of Mean Values for All Species	4.8	5.3
Range of recreationally important species (weighted average)	4.6	5.4

Our review of the empirical effects literature demonstrates the difficulty in discerning a single pH threshold that could ever adequately characterize the ability of a water body to sustain recreational fishing. The most rigorous use of the available data might employ species-specific thresholds and apply these thresholds to individual lakes in the economic modeling domain. Unfortunately, information on the prevalent recreationally important species present in the sample of lakes modeled by MAGIC, as well as in the larger domain of lakes to which these results would be extrapolated, is not currently available. We therefore adopt a range of estimates of a pH threshold for acidification of a lake of 5.0 to 5.4. The range is consistent with a reasonable approximation of effects noticeable to anglers. Knowledge of effects with certainty would imply a more conservative assumption of a lower pH, perhaps consistent with the low end weighted average for recreationally important species, reported in the last row of Table E-14. We therefore report acidification results for an extreme low end threshold estimate of pH 4.6, but do not interpret those results as providing useful central estimates for a study specifically concerned with recreational fishing.

The final step in our analysis is to use an economic model that monetizes the impacts to recreational fishing under each of the acidification scenarios. This involves the selection of an economic model that appropriately covers the effects of acidification of multiple sites over the geographic area that is impacted, and the proper integration of the water quality information. The ideal for this application is a random utility model (RUM) that allows for the substitution among sites and fisheries as water quality parameters change, an essential feature when estimating recreational benefits.

Very few models of this type exist, and fewer cover a region of high acidification that is impacted by the CAAA. Efforts by Englin *et al* (1991), Mullen and Menz (1985), and Morey and Shaw (1990) advance this line of inquiry by relating regional acid deposition to recreational fishing damages, but Montgomery and Needelman (1997) are the first to use direct water quality measures in conjunction with a random utility model[16]. The estimation proceeds in three steps. First, a site-choice model determines the impact of water quality and other lake characteristics on the choice of a fishing site among the set of all potential sites. The model estimates the value of the available set of lakes to each New York resident. In the second step, a model predicts whether a New York resident will choose to fish on a particular day. Third, based on the results of the site-choice and fishing decision models, it is possible to estimate the change in economic welfare caused by altering water quality of the lakes available to New York residents.

We use the Montgomery and Needelman model by inputting MAGIC acidification estimates and simulating the impact on anglers. We do not re-estimate the econometric model's parameters for this application due to resource constraints, though it is important to note that the model was originally estimated to describe angler response to acidity at pH 6.0, while we assume that anglers respond at a lower pH level. Data from MAGIC are input to the model in the form of a percentage estimate of the lakes in the Adirondacks that fall below a chosen pH level. The effect of acidification has a negative impact on the utility of anglers that might wish to use that resource. By simulating the effect on anglers' utility of acidifying a percentage of lakes within a region, the model can compute the economic impact of a specific level of acidification. Subtracting the economic value of fishing at our baseline level of acidification from that which would occur if the CAAA were promulgated provides an estimate of the benefits accrued to recreational fishermen from reducing acidification in the region.

Acidification Results

We summarize the results of our acidification projections from MAGIC in Exhibit 19. Each scenario described in these tables provides an estimate of the percentage of lakes in the Adirondacks likely to suffer from acidification given the deposition and

[16]The Montgomery and Needelman model applies a technique developed by Morey, Rowe, and Watson (1993).

nitrogen saturation parameters assigned to that scenario. The exhibits present the level of acidification expected given a specific threshold level of acidity (pH 4.6, 5.0, or 5.4), covering uncertainty associated with the impacts of acidity on a range of aquatic species.

Table E-15 shows that in the year 2010, the CAAA can be expected to reduce the number of lakes whose pH falls below 4.6 by zero percent, below 5.0 by one to four percent, and the percentage of lakes falling below a pH of 5.4 by five percent. These results are obtained by subtracting the acidification estimates for the year 2010 without the CAAA from the acidification estimate for 2010 with the CAAA. Note that we only compare scenarios with consistent nitrogen saturation parameters. One can not reliably compare the impacts of acidification on a nitrogen-saturated watershed in one deposition scenario with a non-saturated watershed in another deposition scenario.

Table E-15
Acidification Results - 2010

Year	Status of CAAA	Level of N Saturation	Percentage of Lakes Acidified at Selected pH Levels		
			pH 4.6	**pH 5.0**	**pH 5.4**
1990 (Base Year)	No CAAA Regulations Promulgated		0%	5%	20%
2010	With CAAA	No Saturation	0%	2%	18%
		Saturated	0%	5%	17%
	Without CAAA	No Saturation	0%	6%	23%
		Saturated	0%	6%	22%
Range of Benefits from CAAA in 2010			**0%**	**1%-4%**	**5%**

Acid deposition between 1990 and 2010 also contributes to lagged acidification impacts after 2010. MAGIC estimates that a significant amount of acidification between 2010 and 2040 is avoided by the CAA. This is an area that requires further research in order to fully quantify these impacts.

Economic Results

As acidification of the Adirondacks is reduced by the CAAA, economic benefits accrue to society. In annual terms, the economic benefits of the CAAA in 2010 are summarized in Table E-16. Following the presentation used for the acidification data, the range of annual benefits from the CAAA is \$12 million to \$49 million using an effects threshold of pH 5.0, and \$82 to \$88 million for an effects threshold of pH 5.4. These results correspond to previous analyses (Englin et al, 1991; Mullen and Menz, 1985; Morey and Shaw, 1990) that find annual benefits to the Adirondacks of halving utility emissions in the millions to tens of millions of dollars. We do not provide an economic assessment of acidity in 2040, as the behavioral model is not sufficiently robust to estimate economic impacts 50 years into the future.

Table E-16
Annual Economic Impact of Acidification in 2010
(Millions of 1990 Dollars)

Year	**Status of CAAA**	**Level of N Saturation**	**Economic Impact of Acidification at Each pH Threshold**	
			pH 5.0	pH 5.4
1990 (Base Year)	No CAAA Regulations Promulgated		$61	$320
2010	With CAAA	No Saturation	$24	$281
		Saturated	$61	$261
	Without CAAA	No Saturation	$73	$363
		Saturation	$73	$349
Range of CAAA Benefits in 2010			**$12-$49**	**$82-$88**

We calculate the cumulative economic benefits from the CAAA by summing the difference between the discounted annual economic impact of acidification with and without the CAAA for every year from 1990 to 2010.[17] We perform this calculation for the minimum and maximum values for each of the three pH thresholds for survival of recreational fish species, assuming a straight line increase in the level of acidification between 1990 and the 2010 level for each scenario. We present the results as a cumulative net present value calculated with 1990 as the base year (i.e. costs in 2010 are discounted over 20 years). As indicated in Table E-17, the range of cumulative potential benefits from the CAAA between 1990 and 2010 is from $67 to $465 million.

[17] The formula is $\sum (c_t-f_t)/(1+r)^t$
Where: c = economic impact in the baseline, or counterfactual case;
f = economic impact with the CAAA, or the factual case;
t = the year, where 1990 is year 1;
r = the social discount rate, in this case we use 5%.

Table E-17
Cumulative Economic Benefits of Acidification from 1990 to 2010
(Millions of 1990 Dollars)

	Economic Impact of Acidification at Each pH Threshold	
	pH 5.0	**pH 5.4**
CAAA Benefit Minimum	$67	$433
CAAA Benefit Maximum	$271	$465

Avoided Cost of Liming

An additional factor that must be considered in light of the context of economic damages from acidification is the possibility of mitigating these damages by local means. In the case of the Adirondacks, acidic lakes are systematically limed in order to raise pH and improve habitability for recreational fish species, which can be stocked after liming. This alternative is costly for local resource managers, and difficult to conduct in most Adirondack lakes with limited access, but it does serve to locally mitigate the damages caused by acid deposition. Naturally, damaged aquatic ecosystems can not be entirely replaced by liming and restocking, but impacts to recreational fishing may be minimized. Currently a limited number of lakes are limed in the Adirondacks. In this section we examine the economic implications of this practice and demonstrate that liming will be necessary both with and without the CAAA.

The goal of liming in the Adirondacks is to mitigate the effects of acidification by the addition of acid neutralizing products in selected waters to maintain and/or restore brook trout populations. Waters may be considered for liming and re-stocking, if the pH drops below 6.0.[18] The present liming program is limited in scope due to policy constraints, environmental regulations, and factors affecting the economic feasibility. With a few exceptions, lakes are typically not limed, if any of the following applies: the water is considered naturally acidic; the flushing rate is greater than two times a year; the water will not support brook trout regardless of the pH of the water; or, liming of the water would be too expensive, due to its remote location.[19]

It is not possible to predict the number of lakes that the New York Department of Environmental Conservation (NYSDEC) would choose to lime and restock in 2010 based on the available data, but we can estimate the potential costs and impacts if the program remains constant from 1990 to 2010, or if it grows at NYSDEC's proposed rate of two additional lakes per year. In 1990 approximately 25 percent of the region's lakes suffered a pH below 6.0, according to MAGIC, and NYSDEC limed and monitored 32 lakes and restocked 30. MAGIC estimates that the same percentage of lakes will maintain a pH below 6.0 in 2010, both with and without the CAAA. Table E-18 presents the cumulative costs associated with liming lakes in the region from 1990 to 2010.

[18]Personal communications with Larry Straight, Rick Costanza (NYSDEC, Region 5), and Bill Gordon (NYSDEC, Region 6).

[19]NYSDEC, 1990 and personal communications with Larry Straight, Rick Costanza (both at NYSDEC, Region 5), and Bill Gordon (NYSDEC, Region 6).

Table E-18
Cumulative Cost of Ph Stabilization from 1990 to 2010
(Millions of 1990 Dollars)

	Number of Lakes	Cost of Liming	Cost of Monitoring	Cost of Stocking	Total Cost
Program Remains Constant	32	$0.11	$0.07	$0.23	$0.40
Program Grows by Two Lakes per Year	72	$0.16	$0.10	$0.35	$0.61

Under the current plan to lime lakes with a pH below 6.0, this practice will continue under both scenarios of our analysis - with and without the CAAA. Therefore, liming costs will be incurred regardless of regulatory efforts. What we can not determine is the impact that liming may have on our avoided damages analysis. It is possible that liming lakes with the greatest recreational potential will offset the majority of economic impacts of acidification, though due to the structure of our economic modeling approach it is not possible to test this hypothesis at present. On the other hand, it is important to note that liming is feasible only on lakes where access is very easy, and therefore is limited in the scope of its impact. Furthermore, as previously stated, liming is a stop-gap measure that is both temporary and not a complete substitute for restoring natural ecosystem conditions.

Caveats and Uncertainties

The impacts of acid deposition in the eastern United States include both terrestrial and aquatic ecosystem damages. Many of these effects are difficult to measure, and most are impossible to monetize given current methods. The result is that our analysis treats a very narrow definition of the impact of acidification. A far more broad definition would include costs associated with damaging the integrity of terrestrial and aquatic ecosystems, many of which are not quantifiable at this time. Nevertheless, our case study of the Adirondacks region demonstrates that the CAAA is generating substantial economic benefits in just the narrow scope of recreational fishing. Our analysis states that by mitigating the impacts to recreational fisheries from acidification with the regulations promulgated under the CAAA, benefits can be accrued in the hundreds of millions of dollars.

The limitations that affect these estimates are caused by data and computational constraints at each stage of the simulation process. We detail each of these limitations below, and indicate the directional bias these limitations may create in our final benefits estimates.

Emissions, Deposition, and Acidification Estimates

Each of the models that contribute to the acidification estimates (i.e. the emissions model, RADM, and MAGIC) has been rigorously tested. For example, MAGIC estimates of acidification have been tested extensively including the following procedures: individual process formulations in the model have been tested against laboratory experiments with soils; model hindcasts of historical lake chemistries in the Adirondacks have been made and compared with values inferred from lake sediment records; and, predictions of the effects from whole-watershed manipulations have been compared to observed effects. Nevertheless, it is well documented that MAGIC estimates suffer from unquantified uncertainty, parameterization, and validation problems (EPA, 1995).

It is beyond the scope of this report to dissect MAGIC, RADM, or the emissions projections to identify factors that might affect their results. Furthermore, it is not possible to quantify the cumulative uncertainty that propagates in the linking of these models to provide acidification estimates. It is sufficient to note that the acidification estimates generated by MAGIC should be treated with proper caution, applying sensitivity analysis to any further modeling work that uses these data as input. Several limitations are detailed below.

- We consider the potential effects of nitrogen saturation on lake acidification by performing sensitivity analysis of two boundary conditions, representing a nitrogen-saturated watershed and a watershed with complete nitrogen uptake. While this approach allows us to estimate the range of effects nitrogen saturation may have on acidification of surface waters, it does, however, not account for the fact that nitrogen saturation is a continuous process leading to increased leaching of nitrogen compounds as a watershed progresses through the various stages of nitrogen saturation (see for example: Stoddard, 1994).

- The sample of lakes simulated by MAGIC must be extrapolated to the entire population of lakes in the region. In order to simulate the complex hydrological, biological, and chemical dynamics of lakes, intensive data collection is required, forcing the developers of MAGIC to limit the number of simulated lakes to only 33. This sample represents lakes with an ANC of less than 400 microequivalents per liter (μEq/L). Lakes with greater ANC are believed not to be vulnerable to acidification from acid deposition. The results from MAGIC therefore are only applicable to those lakes with ANC less than 400 μEq/L, but we have no assurance that the sample of 33 lakes is representative of the distribution of lake ANC levels below 400 μEq/L in the total population. In addition, we are forced to use pH 7 as a proxy for ANC 400 μEq/L where ANC data is not available. Though pH and ANC are correlated, there is significant variance in this relationship.

Ecological Factors

In the ecological assessment two major limitations arise. It is not possible to address either of these limitations with sensitivity analyses, so it is important to keep in mind that results may be biased by these factors.

- Acidic episodes may significantly affect fish populations. They are, however, not considered in our analysis due to significant limitations of our ability to model episodic events.

- It is well documented that pH is not the only factor that determines fish survival, although we do use it as the single indicator for ecological health. This overlooks the importance of other components of water chemistry such as aluminum and calcium concentrations. This is necessary because there is insufficient data for our geographic region to develop a sufficiently sophisticated ecological-economic model that would consider all these variables. Because we test several pH thresholds at which anglers might perceive declining fish populations, this simplification of ecosystem dynamics should not bias our final economic estimates.

Economic Estimates

The economic model is also subject to some uncertainty that may bias our monetary results. Again, it is not possible to address all of these limitations with sensitivity analyses, so it is important to keep in mind that results may be biased by these factors.

- This analysis includes only day trips to sites within three hours of the angler's home. We take no account of people who would come and spend several days fishing at a site. By excluding these people we likely understate the costs of increased acidity.

- This model treats every day as a potential fishing day, and assumes that each fishing occasion is independent of all others. Neither assumption is realistic. The implication is that our measure of the seasonal costs are likely biased upward.

- Montgomery and Needelman do not compute a confidence interval for the value estimates of lake acidification in their study. This could result in an overestimate of the economic impact of acidification as we can not determine that the value estimates are significantly different from zero.

- Even though this study offers a much more comprehensive set of alternative fishing sites than most recreational fishing studies, we were unable to account for rivers and streams, or for lakes and ponds in nearby states. To the extent that these alternative sites are substitutes for New York lakes, our welfare measures may overstate the costs of acidification.

- We assume that anglers perceive the effects of acidification at a pH threshold lower than that at which the random utility model was estimated. This is a conservative approach which potentially underestimates the total impact of acidification, but overestimates the benefits of the CAAA because the difference between the percentage of lakes that are acidified in our baseline and CAAA scenarios is larger at lower threshold pH levels.

Timber Production Impacts from Tropospheric Ozone

The purpose of this section is to evaluate the prospective benefits of improved commercial timber growth through the reduction of tropospheric ozone concentrations attributable to the CAAA. Tropospheric ozone (O_3) is a secondary pollutant that is created in the atmosphere by a photochemical reaction among nitrogen oxides (NOx) and volatile organic compounds (VOCs). Documented scientific evidence suggests that elevated ozone concentrations in the troposphere disrupt ecosystems by damaging and slowing the growth of vegetation. We examine one aspect of these impacts in this analysis, reduced commercial timber growth, and find the cumulative impacts from 1990 to 2010 to be $1.87 billion.

Ecological Effects of Ozone

In terms of forest productivity and ecosystem diversity, ozone may be the pollutant with the greatest potential for regional-scale forest impacts (NAPAP, 1991). Studies have demonstrated repeatedly that ozone concentrations commonly observed in polluted areas can have substantial impacts on plant function (see U.S.EPA 1996; De Steiguer 1990; Pye 1988 for summaries).

Like carbon dioxide (CO_2) and other gaseous substances, ozone enters plant tissues primarily through apertures in leaves in a process called stomatal uptake. To a lesser extent, ozone can also diffuse directly through surface layers to the plant's interior (Winner and Atkinson 1986). Once ozone reaches the interior of plant cells, as a highly reactive substance, it inhibits or damages essential cellular components and functions, including enzyme activities, lipids, and cellular membranes, disrupting the plant's osmotic (i.e., water) balance and energy utilization patterns (U.S.EPA 1996; Tingey and Taylor 1982). Damage to plants is commonly manifested as stress specific symptoms such as chlorotic or necrotic spots, increased leaf senescence (accelerated leaf aging) and reduced photosynthesis. All these factors reduce a plants' capacity to form carbohydrates (U.S.EPA 1996), which are the primary form of energy storage and transport in plants. Reduction of carbohydrate production and disruption of carbon allocation patterns in turn can impact the growth rates of trees, shrubs, herbaceous vegetation and crops.

In this section we focus on the economic impacts of reducing commercial timber growth on the U.S. economy. Timber supply is a direct ecological service flow affected by tropospheric ozone and is therefore an ideal quantitative example of the benefits of controlling tropospheric ozone in the U.S. Nevertheless, it is important to note that this benefit represents only a small portion of the overall ecological benefits of reducing the impacts of tropospheric ozone on ecosystems across the nation.

Modeling Timber Impacts from Ozone

In this section we describe our methods for quantifying the impacts of tropospheric ozone on commercial timber production. The assessment of the benefits of regulating tropospheric ozone involves three major steps:

- Estimation of ambient ozone concentrations under a regulatory and a non-regulatory scenario;
- Estimation of the growth changes from ozone exposure on commercial forests;
- Estimation of the economic impact of changes in commercial timber growth.

We describe the completion of each of these steps, the models we use and their input data. Upon completion of these steps it is possible to compare the tropospheric ozone concentrations, ecological effects, and resultant economic impacts over the period 1990-2010 both with and without the CAAA. The net difference between ecosystem effects and economic impacts with and without the CAAA represents the benefits accrued to society from the implementation of the CAAA.

Step 1: Estimating Ambient Ozone Concentrations

In order to simulate the impacts of ozone on commercial forest productivity we must estimate the ambient ozone concentrations at which forests are exposed both with and without the regulations of the CAAA. We accomplish this using historical ambient ozone data for 1990, and projected ozone data for the years 2000 and 2010. We use historical hourly ozone concentrations from EPA's Aerometric Information Retrieval System (AIRS).[20] AIRS is a comprehensive database that contains ambient air quality monitor data for the contiguous U.S. To estimate future year concentrations of ozone we use the Urban Airshed Model (UAM-V), a three-dimensional photochemical grid model that calculates concentration of pollutants by simulating the physical and chemical processes in the atmosphere.

Step 2: Ecological Effects

We use the PnET-II model to estimate the impacts on timber growth of elevated ambient ozone. The model assesses the average change in productivity for softwood and hardwood forests in each of nine timber growing regions defined by the U.S. Forest Service (see following section). The strength of PnET II is that it provides a means to use a geographically transferable method to assess forest stand-level estimates of the impacts of ambient ozone on productivity. For the purposes of a national assessment, this provides a significant advantage over alternative existing methods based on plant-level models (e.g. the tree grow model (TREGRO)) or expert opinion surveys (e.g. . Pye et al. 1998; deSteiguer 1990). The disadvantages of the model include: the use of photosynthetic rates as an indicator of ozone impacts rather than a mechanistic measure of respiratory change; potential bias created when scaling net primary productivity (NPP) changes in plants to the forest stand level; and the use of an ozone measure that may be overly sensitive to changes in ambient concentrations (D40[21]). We discuss the major facets of the model's construction below.

The PnET-II model is a monthly time step, canopy- to stand-level model of forest carbon and water balances based on several generalized relationships (e.g. maximum net photosynthesis as a function of foliar nitrogen content). Carbon and water balances are linked in that potential evapotranspiration is determined as a function of leaf gas exchange rates and the atmospheric vapor

[20] See "www.epa.gov/airs",

[21] The D40 measure represents the cumulative ozone dose above a threshold concentration of 40 ppb.

pressure deficit (i.e. humidity). Actual evapotranspiration is determined from a comparison of potential evapotranspiration with available soil water, which is affected by precipitation, snow melt, direct evaporation from canopy surfaces, soil water holding capacity and a fast flow fraction that represents macropore flow to below the rooting zone.

The model simulates a multi-layered forest canopy that includes gradients in available light, specific leaf weight and hence, leaf level carbon gain. Annual, whole-canopy carbon gain is allocated to leaves, wood and roots after calculations for growth and maintenance respiration costs. The model has been successfully validated for forest production and water balances at a number of temperate and boreal forest sites. For a full description of model algorithms, inputs, assumptions, sensitivity analyses and validation exercises, see Aber and Federer (1992), Aber et al,.(1995) and Ollinger et al. (1998).

PnET-II uses an algorithm to allow prediction of ozone effects on forest growth that relates ozone-induced reductions in net photosynthesis to cumulative ozone uptake (Ollinger et al. 1997). Uptake is determined for ozone concentrations above 40 ppb and is affected by ozone exposure levels and leaf gas exchange rates. Application at sites located across the northeastern US show an interesting interaction between ozone and water availability whereby the occurrence of drought stress reduced ozone damage via reductions in stomatal conductance, and hence, ozone uptake.

Step 3: Economic Impacts

To monetize the ecological effects of elevated ambient ozone on commercial timber production it is necessary to estimate the market changes that result from reduced timber growth rates. We use the USDA Forest Service Timber Assessment Market Model (TAMM) to analyze the changes in timber inventories that would result under each of our ozone exposure scenarios, and the consequent changes in harvests, prices and regional demand for timber. Using the inventory and market computations, TAMM estimates the overall economic welfare impact of changes in forest growth rates, in terms of changes in consumer and producer surplus. Previous peer-reviewed EPA analyses of changes in timber productivity (U.S.EPA 1997) use this same model.

There are three stages to the economic estimation. First forest growth rate information generated by PnET-II is provided to the forest inventory tracking component of TAMM, called Aggregate Timber Land Assessment System (ATLAS). Growth rate information is provided for each of the forest production regions defined by TAMM. (We do not simulate changes in the Canadian regions for this analysis.) Second, ATLAS generates an estimate of forest inventories in each major region, which in turn serves as input to the market component of TAMM. In the third stage, TAMM estimates the future harvests and market responses in each region. A detailed description of TAMM's structure is found in Adams and Haynes (1996).

Ecological Results

P-Net II partitions NPP of forest trees according to tissue type. Changes in NPP for wood tissue result in changes in tree growth rates. On the whole, P-Net II estimates that commercial timber growth rates are improved as a result of reduced tropospheric ozone exposure attributable to the CAAA. The improvement in growth rates by the year 2010 ranged from negative 0.56 percent to 10.91 percent. Table E-19 summarizes the estimated changes in growth rates, by region, for the entire U.S.

Table E-19
Difference in Commercial Timber Growth Rates With and Without The CAAA

Region	Difference in 2000		Difference in 2010	
	Softwoods	Hardwoods	Softwoods	Hardwoods
PN W-E	1.68%	1.58%	2.11%	1.25%
PN W-W	1.17%	0.42%	-0.56%	1.13%
S. West	0.84%	1.77%	-0.14%	1.59%
N.Rocky	2.67%	0.40%	4.46%	2.05%
S. Rocky	4.77%	2.25%	4.14%	3.88%
S. Central	4.54%	4.80%	7.93%	8.41%
S. East	5.40%	5.65%	10.38%	10.91%
N. Central	1.80%	5.74%	4.36%	9.22%
N. East	4.27%	6.68%	9.58%	11.49%

It is important to note that the difference in growth rates gradually grows from zero percent in 1990 to the values presented for 2000, and then 2010. In other words, the difference in growth rate estimated for 2010 is not experienced over the entire 1990-2010 modeling period.

Economic Impacts

TAMM estimates that there is a measurable difference in timber harvests attributable to ozone exposure under our two scenarios. At the outset of our modeling period, early 1990s, virtually no change is measured in forest harvest volumes. This is an expected result because increases in growth rates should not substantively affect timber volume over so short a period of time. By the end of our modeling period, late 2000s, increased growth rates over the previous decade(s) begin to affect overall forest yields in the form of harvestable timber. This is observed in Figure E-8 as an increasing annual benefit estimate over the modeling period.

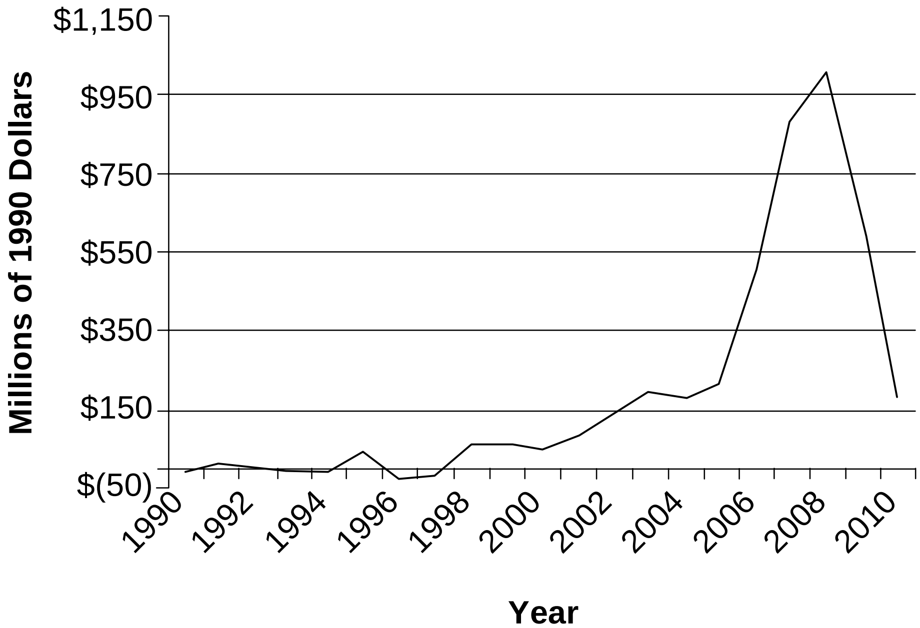
The shape of the benefits time-series reveals a production shift in one region of the United States as a result of increased timber availability. This shift produces a spike in economic surplus for a period of three years. Although this change is small in percentage terms relative to total economic surplus generated by the timber sector it contributes to a large portion of the benefits estimate over the 1990-2010 period.

The cumulative value of annual benefits is calculated as the sum of the annual differences in consumer and producer surplus from commercial timber harvests under our CAAA and no CAAA ozone exposure scenarios from 1990 to 2010. We discount annual benefits to 1990 dollars using a five percent discount rate. The total cumulative benefits estimate is $1.87 billion.

Caveats and Uncertainties

In interpreting results from this analysis, several points should be considered. First, large-scale analyses of complex ecosystem processes are typically conducted with simulation models because it is impossible to conduct large-scale manipulation experiments that would provide similar predictive capabilities. This brings with it inherent uncertainties in that there may be little or no data with which to validate model predictions. In the case of ozone effects on forest production, the absence of controlled, whole-forest fumigation studies across the range of climatic, vegetation and pollution conditions experienced across the U.S. makes it presently impossible to validate all model predictions. In this

Figure E-8
Annual Economic Welfare Benefit of Mitigating Ozone Impacts on Commercial Timber: Difference Between the Pre-CAAA and Post-CAAA Scenarios



analysis, we have combined established empirical relationships between ozone exposure and plant physiological function in a peer-reviewed model that is based on sound forest growth processes. As such, the resulting model predictions should be viewed as a set of refined hypotheses, but nevertheless, hypotheses that have not been thoroughly tested.

Second, while ozone has repeatedly been identified as an important environmental stress agent affecting forest vegetation, it is not the only such factor to which forests are currently exposed at regional to global scales. Human activities have profoundly affected global cycles of carbon, nitrogen and a number of other elements in ways that may be at least as important as ozone. Because a number of changes (e.g. elevated CO_2 and increased atmospheric nitrogen deposition) have significant potential to cause large-scale fertilization effects, growth predictions that include ozone effects alone should be viewed as incomplete.

Ozone Modeling

- Because it is not possible to model ozone levels throughout the country during the months of October through April during future years, it is necessary to employ another method to obtain estimates for ozone levels during these months. We assume that ozone levels during these months for 2000 and 2010 will be identical to the levels during the same months of 1990. Thus, any differences in timber production under the two scenarios of CAAA promulgation and no CAAA promulgation will be driven solely by ozone differences during the warmer part of the year that comprises the majority of the growing season.

- It is important to note that ozone monitoring is not complete, with coverage especially low in forested regions of the United States.

Only two percent of ozone monitors are in forested areas (U.S.EPA 1996). We work with the best possible estimates of tropospheric ozone concentrations but identify this as a significant area of uncertainty in this analysis.

Ecological Modeling

- Preliminary model results revealed an interesting and unexpected interaction between ozone, drought stress and carbon allocation. On moist, productive sites, ozone resulted primarily in decreased wood growth because the simulated trees can afford to lose wood without reducing more important tissues which are given higher allocation priority in the model (leaf and root). On progressively colder or drier sites, ozone exposure causes reductions in all plant tissues because plants are already stressed enough that additional reductions in carbon gain must come from all plant pools. Complex interactions among ozone, drought and this carbon allocation dynamic produced unexpectedly variable results, which, in some cases caused an increase in growth in response to ozone. Although these are interesting and biologically feasible interactions, in the absence of any real data in this area, it is impossible to determine the extent to which they actually occur.

Economic Modeling

- There are two important caveats to the economic modeling. First, we generalize changes in growth rates for entire forest types across potentially heterogeneous regions. TAMM is capable of modeling timber growth and harvest with greater precision, breaking down forests into many species and age-classes and by county. We do not anticipate that increasing the precision of growth rate data on a national scale would substantially alter our results.
- The second caveat is economic benefits may be underestimated by using so short a modeling period. It is evident from the data we present that improved growth takes years to affect actual harvests. Therefore, the complete benefits of improved growth during 1990 to 2010 will not be accrued until after 2010. By not including these years in our analysis we can not fully account for the commercial timber benefits of ozone mitigation over the period of our analysis.

Carbon Sequestration Effects

It is possible to extend the analysis of timber growth rates to account for the differences in temporary and long-term carbon sequestration under each of our ozone scenarios. This is accomplished by linking two USDA Forest Service Models to TAMM/ATLAS to generate estimates of carbon sequestered in standing forest, and carbon sequestered in commercial forest products. We briefly summarize those steps here.

TAMM/ATLAS provides an estimate of the standing timber stock and the commercial timber harvests that will occur under each of our ozone exposure scenarios. Using this information we estimate the respective volumes of carbon sequestered in each scenario using the forest carbon model (FORCARB) and harvested carbon model (HARVCARB).

FORCARB contains a set of stand level carbon budgets that relate the timber growth and yield output from ATLAS to trends in total ecosystem carbon over the course of stand development. These include carbon sequestered in trees, woody debris, understory vegetation, and the forest floor. Using these data, FORCARB estimates the total carbon sequestered in commercial forests at any point in time. This information provides a useful baseline for the rate of forest carbon sequestration that can be expected under different ozone exposure scenarios. For a complete description of FORCARB and its application see Turner et al. (1993) and Turner et al. (1995).

The age of natural forests and the management regime of commercial forests largely determine the fate of forest carbon. In natural forests, carbon sequestration is temporary, with sequestered carbon eventually returning to the nutrient cycle. Alternatively, harvested timber is transformed into commercial products that alter the life cycle of sequestered carbon. Using the HARVCARB model, we use harvest information from TAMM to track the lifecycle of timber. The ultimate fate of this sequestered carbon depends on the efficiency of timber conversion (i.e. how much timber becomes a product), and the durability of that product. HARVCARB estimates the long-term carbon sequestration resulting from timber harvests under each of our scenarios. A full description of HARVCARB is found in Row and Phelps (1990).

Forest ecosystems help mitigate increasing anthropogenic carbon dioxide emissions by sequestering carbon from the atmosphere, converting atmospheric carbon into biological structures or substances needed in physiological processes. Some air pollutants, however, may adversely affect the potential of forests to sequester carbon by slowing down the rate of biomass accumulation of sensitive forest tree species. This may affect the global carbon cycle and may contribute to anthropogenically induced changes in the earth's climatic conditions.

Using output from TAMM/ATLAS, timber inventories can be converted into estimates of carbon sequestered in commercial forests by a forest carbon model (FORCARB). FORCARB estimates the carbon storage in each of four ecosystem components: trees; forest understory, forest floor, and soil. The model uses forest carbon storage and flux estimates based on ecological analyses of each of the forest ecosystem components. The details of these studies and their synthesis into the FORCARB model can be found in Birdsey (1992a, 1992b) and Heath and Birdsey (1993). Heath and Birdsey (1995) provide a technical description of integrated simulations using TAMM/ATLAS and FORCARB. Of the carbon sequestered in forests, some portion is subsequently harvested as timber and processed into wood products, paper, and biomass fuel. We use a harvest carbon model (HARVCARB) to estimate the life-cycle of harvested forest timber and thereby adjust the forest carbon sequestration estimates of FORCARB. HARVCARB relies on a range of assumptions approximately 50 percent of harvested wood ultimately becomes a wood or paper product, the remainder becomes waste from the production process. Of the final wood and paper products, a small percentage become durable products or are landfilled and decompose at a rate of less than one percent a year (Row and Phelps, 1990). Wood that is either manufactured into a durable product (e.g. permanent building construction material, furniture) or materials that are landfilled (e.g. paper) contribute to long-term carbon sequestration. The remainder of the harvested wood mass (e.g. biomass fuel, non-durables that are not landfilled) is re-released to the environment and therefore is not included in the volume of carbon estimated to be sequestered in forests.

We find that forest carbon sequestration increases with improved air quality under the CAAA. This result corresponds with the intuition that forests tend to grow faster when tropospheric ozone exposure is reduced. Carbon flux, or annual forest carbon sequestration minus forest harvest losses (excluding long-term carbon sequestration in forest products) is also greater under the CAAA than under our No-CAAA air quality scenario. We summarize our results in Table E-20.

Table E-20
Differences in Carbon Flux (millions of metric tons/year)

	1990-1999	2000-2010
Forest Flux	8	28
Land Use Change	> -1	> -1
Cumulative Fate of Removals	> 1	> 1
TOTAL FLUX	**8**	**29**

Forest carbon flux attributable to the CAAA represents approximately four to sixteen percent of anticipated total carbon flux in U.S. forests between 1990-2010.

In the event of a binding international carbon mitigation agreement, the implication of this result is that substantial costs of carbon mitigation may be avoided by improved forest growth attributable to the CAAA. Though it is not possible to evaluate the monetary value of the avoided cost at this time due to uncertainty regarding the actual cost of carbon mitigation, it will be possible to estimate the value using the data in this analysis once reliable carbon mitigation costs become available.

Caveats and Uncertainties

Additional caveats and uncertainties associated with the estimation of carbon sequestration in U.S. commercial forests include the following:

- FORCARB estimates are based on a synthesis of a variety of empirical studies of the four ecosystem components (soil, forest floor, understory, and trees). The total error of the composite of these studies is not treated explicitly as a modeling output.

- FORCARB also estimates the carbon storage and flux for a variety of forest types based on a synthesis of empirical studies. The error associated with extrapolating these data across a variety of forest ecosystem types is not explicitly treated.

- HARVCARB utilizes data on the life span of durable wood products that is over 50 years old, originally compiled by the Internal Revenue Service for purposes of calculating depreciation of these products. Though the authors of HARVCARB state that this data continues to be reliable, changes in construction, product and their uses most likely biases these data. No estimate is made of the magnitude or direction of this bias.

Aesthetic Degradation of Forests

The purpose of this section is to evaluate the prospective benefits of forest aesthetic improvements associated with improved air quality attributable to the CAAA. In order to assess these benefits, we first evaluate the known changes in visible injuries over time. Available scientific methods and data on the visual appearance of forest stands and their impact on perceived forest aesthetics, however, make it difficult to precisely describe changes in forest aesthetics. Nevertheless, it is possible to describe a range of visual impacts that may be caused by air pollutants and their potential effect on forest aesthetics. Second, we assess the economic value associated with such aesthetic changes. The focus of much of this work tends to be site-specific, describes the aesthetic impacts of a number of causal factors, and utilizes a variety of experimental methods making it difficult to generalize results. We conclude that air quality improvements attributable to the CAAA should result in improved forest health, possibly providing aesthetic value to society in the range of billions of dollars. A more detailed description of this analysis is found in *Characterizing the Forest Aesthetics Benefits Attributable to the 1990 Clean Air Act Amendments, 1990-2010* (IEC, 1999c).

Forest Aesthetic Effects from Air Pollutants

Air pollution can cause a wide variety of visual injuries to forest stands, ranging in severity from subtle injuries (e.g., minor leaf discoloration) to severe forest decline (e.g., extensive defoliation and death of trees). The severity of symptoms depends on many factors including the atmospheric concentration of air pollutants, the sensitivity of tree species to air pollution and the presence of other environmental stress factors (Fox and Mickler, 1995; Eagar and Adams, 1992; Olson et al., 1992; Smith, 1990).

Many CAAA-regulated air pollutants are associated with visual symptoms, including, but not limited to, tropospheric ozone, sulfur dioxide and hydrogen fluoride, the three major pollutants known to have caused significant visual injuries to forest trees in the past (NAPAP, 1987). Other air pollutants known to potentially cause visual injuries to plants are strong mineral acids, precursors of which are also regulated by the CAAA (NAPAP, 1987). In addition, there are a variety of other air pollutants potentially affecting the visual appearance of plants, including heavy metals such as lead and mercury (EPA, 1997d; Gawel et al., 1996; Smith, 1990; NAPAP, 1987); nitrogen oxides; ammonia; peroxyacetyl nitrate; chlorides; and ethylene (Smith, 1990; NAPAP, 1987; Jacobson and Hill, 1970). However, very limited information is presently available on visual damages caused by these pollutants. Tables E-21 and E-22 summarize the known visual impacts of air pollutants on forests and their geographic extent.

As a consequence of complex natural forest dynamics, lack of extensive long-term monitoring networks, and difficulties in establishing cause and effect relationships, it is not possible to quantify the extent of visual forest injuries caused by air pollutants or changes that may have occurred since the implementation of the CAAA. In addition, mechanisms that induce threats to forests may operate on such a long-term scale that benefits in the visual appearance of forests may not be exhibited during the period of our analysis.

Table E-21
Typical Impacts of Specific Pollutants on the Visual Quality of Forests

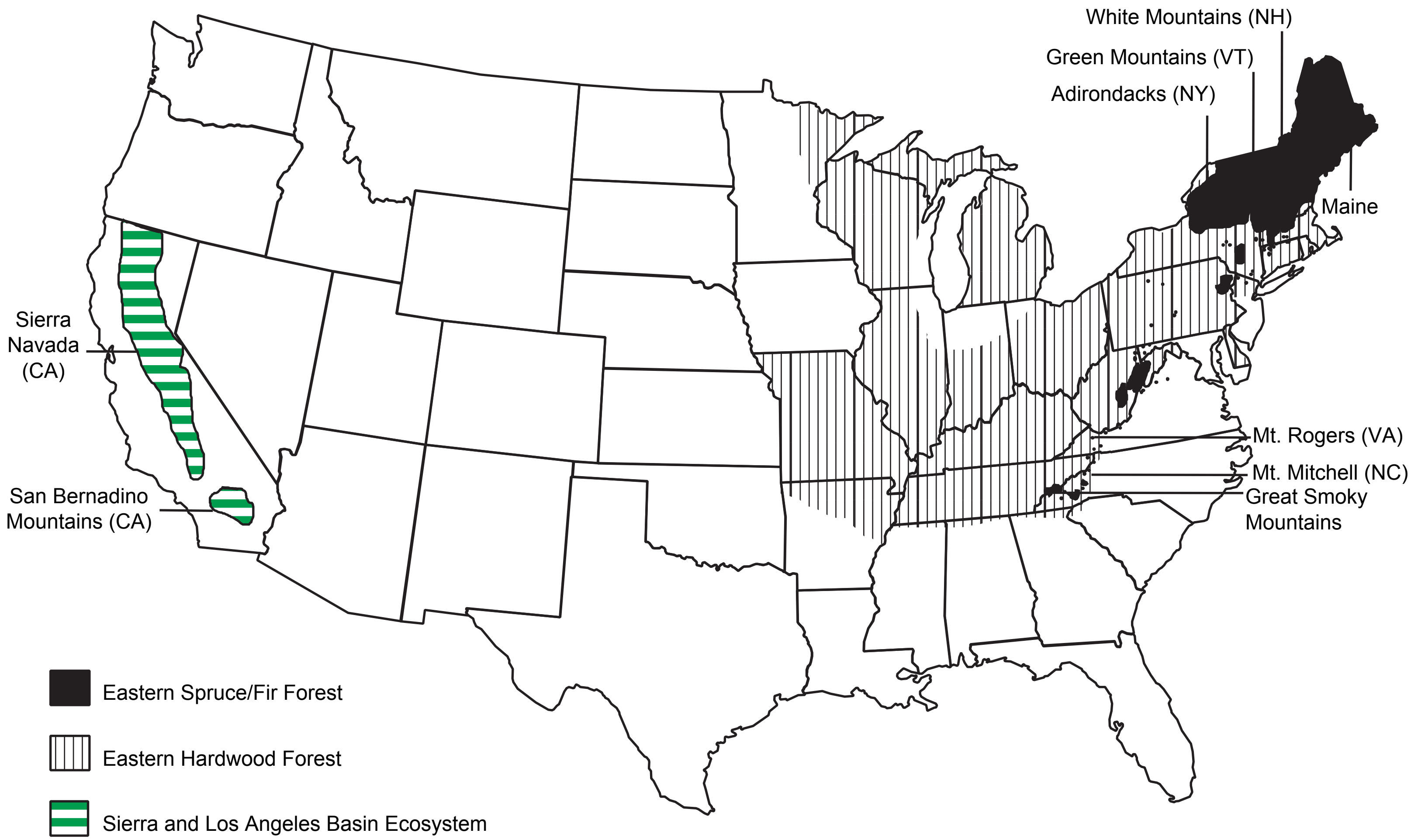
	Geographic Extent	Direct/Indirect Injuries	Major Types of Visual Injuries
Ozone	Area or regional effects	Direct injuries	Foliar injuries (e.g., pigmented stipple), increased needle/leaf abscission, premature senescence of leaves. Pattern, size, location, and shape of foliar injuries to indicator species can be specific for ozone.
		Indirect Injuries	Increased susceptibility to visual injuries that may result from other adverse environmental factors, such as insect attacks. For example, increased needle/leaf abscission, elevated mortality rates, and/or changes in species composition.
Acidic Deposition	Area or regional effects	Indirect Injuries	Increased susceptibility to visual injuries that may result from other adverse environmental factors, such as climatic factors. For example, increased needle/leaf abscission, elevated mortality rates, and/or changes in species composition. Acidic deposition can also cause direct foliar injuries. Acids are, however, more likely to indirectly affect the visual appearance of forest trees, unless exposure levels are very high.
Sulfur Dioxide	Point source pollution	Direct Injuries	Foliar injuries including leaf/needle discoloration and necrosis. Pattern, size, location, and shape of foliar injuries to indicator species can be specific for sulfur dioxide. At high concentrations, elevated mortality rates of sensitive species and changes in species composition may occur. Sulfur dioxide may also cause indirect injuries. Indirect injuries, however, are not well documented.
Hydrogen Fluoride	Point source pollution	Direct Injuries	Foliar injuries including leaf/needle discoloration and necrosis. Pattern, size, location, and shape of foliar injuries to indicator species can be specific for sulfur dioxide. At high concentrations, elevated mortality rates of sensitive species and changes in species composition may occur. Hydrogen fluoride may also cause indirect injuries. Indirect injuries, however, are not well documented.

Table E-22
Forests Affected by Regional Pollution

Affected Forest Type / Species	Region	Major Air Pollutants	Documented Visual Injuries	Suspected Mechanisms of Injury	Sources
Mixed Conifer Forest / Ponderosa and Jeffrey Pines	San Bernardino Mountains, California	Ozone and nitrogen containing substances	Foliar injuries include chlorotic mottle, tip necrosis, premature senescence of needles, and increased needle abscission. Elevated mortality rates and changes in species composition have occurred.	Direct ozone-induced foliar injuries. Heavy bark beetle attacks facilitated by drought, ozone, and nitrogen containing air pollutants. Ponderosa and Jeffrey pine have shown air pollution-related symptoms of decline probably since the mid 1950s.	E PA 1996a; Miller, 1992; Stolte et al., 1992; NAPAP, 1991; Miller and McBride, 1998
	Sierra Nevada, California	Ozone	Foliar injuries include chlorotic mottle, tip necrosis, premature senescence of needles, and increased needle abscission.	Direct ozone-induced foliar injuries. The Sierra Nevada contains the largest forest area in the world with documented damage from a non-point source pollutant but ozone exposure and injuries are not as severe as in the San Bernardino Mountains. Visible ozone-induced foliar injuries were first documented in the early 1970s.	EPA, 1996a; Peterson and Arbaugh, 1992; NAPAP, 1991; Miller and Millecan, 1971

Affected Forest Type / Species	Region	Major Air Pollutants	Documented Visual Injuries	Suspected Mechanisms of Injury	Sources
Spruce-Fir Forest / Red Spruce	High elevation areas in the northern Appalachians.	Acidic deposition (esp. acidic cloud water), and ozone	Foliar dieback, bud injury, foliar loss, Elevated mortality rates.	Acidic deposition increases the susceptibility of red spruce to winter injury (freezing). A dramatic increase in the frequency of winter injury in red spruce stands occurred in the late 1950s and 1960s, coincident with a significant increase in the emissions of precursors of acidic deposition.	EPA, 1995a; Johnson et al., 1992; DeHayes, 1992; NAPAP, 1991
	High elevation areas in the southern Appalachians.	Acidic deposition (esp. acidic cloud water) and ozone	Crown thinning and pockets of high red spruce mortality have been detected on a few mountain sites. Ozone-induced foliar injury.	Acidic deposition leads to nutrient imbalances through accelerated foliar leaching and soil acidification. Soil acidification is characterized by a loss of soil nutrient cations and occurrence of toxic aluminum levels. Also: direct foliar injuries caused by ozone.	EPA, 1995a; Johnson et al., 1992; Johnson and Fernandez, 1992; Cook and Zedaker, 1992; NAPAP, 1991
Eastern Hardwood Forest / Sugar Maple	Northeastern US and Canada	Acidic deposition and ozone	Crown thinning, branch dieback, elevated mortality rates	Acidic deposition leads to nutrient imbalances through accelerated foliar leaching and soil acidification. Soil acidification is characterized by a loss of soil nutrient cations and occurrence of toxic aluminum levels. During 1980s sugar maple declined in many stands in the northeastern US and Canada. Involvement of acidic deposition in sugar maple decline has not been demonstrated but cannot be ruled out.	USFS, 1995b; EPA, 1995a; NAPAP, 1991

Figure E-9
U.S. Major Forest Types Affected by Air Pollution-Induced Visual Injuries



Note: Only areas affected by non-point pollution are shown. Scientific certainty varies with location. Direct ozone-induced injuries also occur in several other locations not indicated (e.g., Southern Forests, Berraug et al, 1995).
Sources: NAPAP, 1991 and White and Cogbill, 1992.

Despite limitations in detecting trends in forest health and associated causal agents, it is possible to identify areas in the US that contain forests known or suspected to experience visual injuries. Forests affected by high concentrations of air pollutants in the vicinity of point sources may provide useful case studies because cause and effect relationships may be easier to establish and visual injuries can be severe enough to cause significant aesthetic impacts. In particular, point sources can lead to well-defined concentration gradients in the prevailing downwind direction causing corresponding gradients of visual injuries (Smith, 1990; NAPAP, 1987).

In contrast, concentrations of regionally distributed air pollutants (e. g., ozone and acidic deposition), can be fairly uniform over large geographic areas. Visual symptoms can be more intense in the vicinity of urban areas or industrial sites but may not be limited to these regions (NAPAP, 1987) making it more difficult to establish cause-and effect relationships. Despite difficulties in establishing cause-and-effect relationships, all identified forest ecosystems likely to have experienced air pollution-induced visual injuries in recent history are affected by regionally distributed air pollutants.

Economic Value of Changes in Forest Aesthetics

Though studies that attempt to estimate the value of changing aesthetics are limited in number and scope, they do suggest that people value forest aesthetics and change outdoor recreational behavior according to the quality of forest health in recreational areas. The shear volume of forest-based recreation in the United States suggests that improvements in forest aesthetics could result in substantial benefits. For example, the United States Forest Service reports that recreation visitor days to national forests have increased over the last ten years from 250 million to over 350 million. With the potential magnitude of aggregated individual preferences in mind, we review several studies that relate individual preference for forests with respect to overall appearance and attempt to extend these analyses to those regions where forests are most affected by air pollution.

Peterson *et al.* (1987) used the contingent valuation (CV) method and a hedonic property valuation model to estimate willingness to pay to avoid ozone-induced forest damage in the Los Angeles area. This contingent valuation survey involved two samples: one made up of recreationalists in the greater Los Angeles area, and the second made up of individuals who owned property within the boundaries of the San Bernardino and Angeles National Forests. Each group was shown a set of photographs depicting varying degrees of vegetative damage. Mean WTP by recreationalists and residents were found to be approximately $43 and $137 per household per year, respectively. The hedonic analysis revealed a significant and positive WTP to avoid homes located in forested areas exhibiting ozone damage. Using these two methods, total damages resulting from the current levels of ozone induced forest injury were estimated to be between $31 and $161 million per year. The study authors rejected a significant percentage of responses as "protest" or "inconsistent" bids (40 percent), which would indicate that many respondents may not have understood or accepted the scenario and the commodity being valued. Apart from this, the study also does not address a series of concerns related to the application of CV to assess nonuse values. First, the survey instrument did not include reminders of budget constraints or substitute goods and services. Second, the survey did not clearly define the commodity. The WTP scenario did not clearly indicate how forest damages were to be mitigated.

Walsh *et al.* (1990) interviewed 200 individuals representing the general population of Colorado and were shown three color photographs representing three levels of forest quality. The mid-level quality was said to represent the present state of the forest (100 to 125 live trees measuring more than six inches in diameter at breast height (dbh) per acre). Respondents were asked their WTP to prevent the lowest state (zero to 50 live trees measuring more than six inches dbh per acre) and attain the highest state (125 to 175 trees per acre in this size class). All respondents were informed beforehand that the damage being valued was due to pine beetle and spruce budworm infestations. Mean WTP per respondent was estimated to be $47. An evaluation of the Walsh et al (1990) study reveals several notable strengths. The survey included reminders of budget constraints, and the authors ensured that respondents were familiar with the commodity being valued and were accustomed to paying for access to recreation sites with good forest quality. Only five percent of the responses were rejected as "protest" or "large" bids. Weaknesses of the study include a small sample size (198), inconsistency between results solicited using different question formats (iterative bidding vs. direct question), and potential biases attributable to framing the question as one of the most important issues affecting Colorado residents and the possibility of a "warm glow" affect concerning payment for a social cause.

Holmes et al (1992) used a CV survey to determine WTP to protect threatened spruce-fir forests in Southern Appalachia from insect and air pollution damage. In this study, residents within 500 miles of Asheville, NC were surveyed about their willingness to pay to eliminate damages to regional spruce-fir forests. The authors used two survey formats, discrete choice and payment cards. The mean willingness to pay for protecting the spruce-fir forests was $20.86 using the payment card method, and $99.57 using the discrete choice method. The study ensured that the sample had adequate knowledge of the commodity being valued, and the overall sample size was large. Unfortunately, several weaknesses arise from the fact that the sample was divided into two groups in order to test different survey formats. The study used a small sample size for each of the tested methods (232 and 236, respectively). The number of protest bids was small (7 to 10 percent), indicating that the respondents understood the function of the survey, but the final results generated by the two different methods were substantially different. This study was later revised in Holmes and Kramer (1996), where the results were published as mean willingness to pay of $36.22 for forest users, and $10.37 for nonusers.

Extending Economic Estimates to a Broader Area

These studies provide an incomplete picture of the total benefits that could be obtained by eliminating visual damages to forests associated with air pollution in the country. As an illustrative calculation, we extend the range of valuation estimates provided in Peterson (1987); Walsh et al. (1990); and Holmes and Kramer (1996) to the major regions of affected landscape in the United States. We do not estimate aesthetic value as a function of forest damage from varying levels of air pollution, but rather provide an estimate of the values placed on avoiding damages characteristically experienced during the 1980s in the United States.

In Table E-23 we present the results from the three studies. We base our calculations of benefits on the value per household of avoiding forest damages multiplied by the number of households in the study region.

In Table E-24 we present the results of an illustrative calculation that extends the "market" for this commodity to a broader group of households. The annual value of avoiding the forest damages is the product of the range of household values in Table E-23 and the total number of households in the states most affected by air pollution.

Table E-23
Summary of Monetized Estimates of the Annual Value of Forest Quality Changes

Study	Aesthetic Change Valued	Value of Change per Household (Current Dollars)	Value of Change per Household (1990 Dollars)[i]	Total Annual Value of Change for Region (Current Dollars)	Total Annual Value of Change for Region (1990 Dollars)[i]
Peterson et al. (1987)	Ozone damage to San Bernardino and Angeles National Forests	$6.31-$32.70[ii]	$7.26-$37.62	$27-$140 million	$31-$161million
Walsh et al. (1990)	Visual damage to Colorado's Front Range	$47	$61.68	$55.7 million	$73.09 million
Holmes and Kramer (1996)	Visual damage to spruce-fir forests in southern Appalachia	$10.81 nonusers $36.22 users	$10.37 nonusers $34.76 users	NA	NA

Note: i.) Values adjusted using all item Consumer Price Index, Economic Report of the President, 1998. Years for current dollar estimates: Peterson et al, 1987; Walsh et al, 1983; Holmes et al, 1991.
ii) Based on 4.3 million households in Los Angeles, Orange, and San Bernardino counties.
iii) Assumes 2.5 million households in North Carolina and 1.8 million in Tennessee.

Table E-24
Illustrative Value of Avoiding Forest Damage in the United States
(1990 Dollars)

Affected System	States Included	Value per Household	Households[i]	Estimated Total Annual Value[ii]	Cumulative Value (1990-2010)[iii]
Sierra Nevada and Los Angeles Basin	CA	$7.26-$37.62	10.4 million	$75.5 million -$391.2 million	$1.02 billion - $5.27 billion
Eastern Spruce Fir and Selected Eastern Hardwood	ME, VT, NH, MA, NY, PA, WV, TN, KY, NC, VA	$7.26-$37.62	23.2 million	$168 million - $872.8 million	$2.27 billion - $11.75 billion

Notes: i.) Household data from 1990 Census; ii) Total Value = Households x Value per Household; iii)Assumes a 5 percent real discount rate.

The results of existing work in this area suggest that improvements in air pollution controls result in positive changes in the aesthetic quality of forest stands. Pollutant control provisions of the 1970 CAA and the 1977 CAAA, for example, may have resulted in a significant decrease or elimination of forests visually affected in the vicinity of emission sources. Further reductions in air pollution emissions mandated by the 1990 CAAA should result in additional improvements in forest health and associated economic benefits derived from improved forest aesthetics.

Our illustrative calculation of the regional effects of improving the aesthetic quality of forest stands (Table E-24) likely overstates the extent of market for this commodity. Estimates presented in Table E-23, however, based on a more conservative application of the extent of market for this commodity, provide a better basis to estimating the order of magnitude of this category of effects of air pollution on ecosystem health. Considering only the Peterson et al. and Walsh et al. studies, conducted in two areas that have been shown in previous assessments to be affected by accumulated air pollution damages, estimates of the total annual value of improvements in the aesthetic quality of forests are in the $100 million to $250 million range.

Caveats and Uncertainties

To quantitatively assess the effects of air pollution emission reductions on forest aesthetic benefits, considerable amounts of high-quality data are required. These data include extensive long-term monitoring networks producing consistent and comparable information over time frames as long as several decades. In addition, injuries captured by monitoring networks have to be linked to the causal agent(s), a task that is currently associated with high factors of uncertainty. Only rarely, if ever, is air pollution the only factor negatively affecting forest health. Typically, a variety of adverse environmental factors act synergistically to induce injuries, considerably limiting our ability to detect air pollution as one of the factors causing injury and to quantitatively assess the amount of injuries attributable to air pollutants.

There are caveats to the use of benefits transfer in this context. The application of this method is intended to provide an order of magnitude estimate of the benefits associated with avoided aesthetic damages to forests in the United States. More sophisticated estimation methods will be required if a truly accurate estimate of value, especially the marginal value of incremental changes, is to be derived. Following is a summary of the caveats to using this approach.

- The impacts that we value are not equivalent to those avoided through the implementation of the CAAA, they are historical effects. A comprehensive assessment of forest aesthetics-related benefits associated with improvements in air quality is limited by significant factors of uncertainty occurring in both the natural science component of the assessment and the economic analysis. Factors of uncertainty in natural sciences include difficulties detecting trends in forest health in general, attributing changes in forest health to specific factors such as air pollution, and establishing valid dose-response relationships of forest exposure to air pollutants and resulting visual injuries.

- The types of aesthetic deterioration in the original studies are not necessarily the same as those experienced in other regions. The nature of forest aesthetic deterioration will vary (e.g. the yellowing of conifer needles vs. gypsy moth defoliation of hardwoods) as will the intensity.

- We do not fully assess the range of potential substitutes for the aesthetic health of regional forests to each household. Having ready substitutes could lower the value a specific household might place on aesthetic quality of regional forests.

- The distinction between marginal values for forest health and average value is not made. As marginal values for changes in forest health diverge from the assumed average value in this analysis, the estimates develop bias.

- We assume that differences in average regional income do not affect estimates.

Toxification of Freshwater Fisheries

The purpose of this section is to assess, from 1990 through 2010, the ecological benefits likely to accrue as a consequence of reductions in the emissions of hazardous air pollutants (HAPs), as mandated by the CAAA. Title III of the CAAA lists 189 chemicals considered to be HAPs. Ideally, a comprehensive economic analysis of the ecological benefits of CAAA-mandated reductions in HAP emissions would include analyses for all service flows potentially affected by the emissions of HAPs. However, a broad quantitative analysis of all these benefits is not yet scientifically possible. What is possible is a qualitative analysis of the likely benefits of reduced HAP emissions for recreational fishing. A more detailed description of this analysis is found in *Economic Benefits of Decreased Air Toxics Deposition Attributable to the 1990 Clean Air Act Amendments*, 1990-2010 (IEc 1998d).

Impacts of Toxic Air Emissions

Five HAPs, mercury, PCBs, chlordane, dioxins, and DDT were responsible for nearly 95 percent of the fishing advisories extant in 1995 (EPA 1996b). The use of three of these compounds (PCBs, chlordane, and DDT) was effectively illegal in the United States prior to 1990 (EPA 1992a), and there are currently no plans for additional CAAA regulations of these compounds (Federal Register Unified Agenda 1998). The remaining two HAPs, mercury and dioxins, are therefore the focus of this analysis.

Because the ecosystem responses to toxic contamination are poorly understood, and observable service flow impacts are difficult to model, we use fishing advisories as a measure of the extent of toxic contamination. In addition, we can characterize the economic impact of HAPs emissions based on altered fishing behavior caused by toxic contamination of freshwater fisheries. It is important to note that fishing advisories alone do not provide a comprehensive view of impacts of toxic contamination on ecosystems, and more expansive measures should be examined in future research.

Fishing advisories are issued by state and tribal agencies when the levels of toxins in the tissue of fish exceed limits established by both state and federal authorities. Fishing advisories generally take one of four forms:

- Advisory for any consumption by the general population;
- Advisory for pregnant women, nursing mothers, and children;
- Advisory for limitation on consumption based on size of fish and frequency of consumption; and
- Advisory for limitation on consumption for specific sub-populations.

According to the U.S. Fish and Wildlife Service (1998), the total number of advisories in the U.S. in 1997 was 2,299, increasing five percent from 1996. The number of water bodies under advisory represents 16.5 percent of the nation's total lake acres and 8.2 percent of total river miles. In addition, 100 percent of the Great Lakes waters and their connecting waters and a large portion of the nation's coastal waters are also under advisory. The total number of advisories in the U.S. has steadily increased for mercury and dioxin.

Mercury is responsible for approximately 75 percent of all fish consumption advisories in effect in 1995 (EPA 1996b). Mercury from point sources, as opposed to mercury deposited from the atmosphere, may be responsible for many of these advisories. The lakes and streams with advisories are concentrated in the northern portions of Minnesota and Wisconsin, as well as in Florida, Missouri, Indiana, Ohio, North Carolina and New England (EPA 1997a, 1997d). Judging by fish consumption advisories, fish mercury levels do not appear to be a widespread problem in the remainder of the United States, and EPA (1997d) found that the typical consumer eating purchased fish is not at risk of methylmercury poisoning. Approximately three percent of fish consumption advisories in effect in 1995 were due to the presence of dioxins (EPA 1996b), and in 1996, 18 states had one or more water bodies under advisement because of dioxin levels in fish (EPA 1997a). Dioxins from point sources may be responsible for many of these advisories.

Several limitations to the fish advisory data exist. First, many lakes, rivers and streams have not been analyzed for toxicity, and it is possible that advisories eventually will be issued for these water bodies. Table E-25 summarizes the sampling intensity for toxicity through 1997. Second, current levels of toxics in watersheds may result in future toxification of healthy water bodies, even in the absence of additional future HAP deposition. Therefore, the current set of fish advisories underestimates the magnitude of toxification from air deposition to date. Third, the protocol for fishing advisory issuance may vary from

Table E-25
Summary of National Data on Toxicity Sampling for Fishing Advisories

Water Body	Percentage of Water Bodies Assessed for Contaminants	Percentage of Assessed Water Bodies Under Advisory
Lakes (acres)	11.36	78.61
Rivers and Streams (miles)	2.41	29.58

Source: EPA 1997a

state to state, removing any consistent basis on which to judge the levels and causes of fisheries' toxicity for each state.

Illustration of Economic Cost to Anglers

The economic welfare implication of water quality changes to recreational fishing are well studied. Most literature in this field focuses on the impacts of deteriorating water quality in a specific fishery. More recently, economic models are appearing that address the social welfare cost of water quality deterioration in multiple fisheries within a region. Such an approach accounts for choices made by fishermen concerning travel to, and the attributes of (e.g., fish advisories), multiple fisheries. Random utility models (RUM) provide the computational method for these regional analyses.

Montgomery and Needelman (1997) were the first to use direct water quality measures in conjunction with a RUM approach to analyze the economic impacts of toxification on regional anglers. Using data from the New York Department of Environmental Conservation (NYDEC), Montgomery and Needelman identify 23 water bodies with toxicity advisories among 2,561 lakes and ponds in the state. Using water quality data and geographic location of both water bodies and anglers, the authors estimate the economic cost of the toxification within the state. The results are presented in Table E-26.

Table E-26
Estimates of the Welfare Cost of Toxification in New York State (1990 Dollars)

Level of Toxicity	Compensating Variation per Trip	Compensating Variation per Capita per Day	Compensating Variation per Capita per Season
Toxic Contamination	$1.23	$0.37	$51.51
Site Closed Due to Toxic Contamination	$1.69	$0.50	$70.92

Source: Montgomery and Needelman 1997

The results from Montgomery and Needelman indicate that the economic welfare implication of existing toxic contamination is substantial for New York State, as described below:

> \$0.37/person/day x 17,990,000 people x 140 fishing days/season = \$931,882,000/season.

In perpetuity,[22] the value of eliminating toxicity in New York State, using a five percent discount rate, is calculated to be \$18,637,640,000.

Clearly, the results using these assumptions are very large. Applications of this model for purposes of estimating the effects of air toxics deposition on recreational fishing requires further investigation of the assumptions in this model.

Jakus et al. (1997) conducted a similar RUM analysis of toxification of reservoirs in Tennessee. Data from the Tennessee Valley Authority showed fishing advisories for two of 14 reservoirs in central Tennessee, and six of 14 reservoirs in the eastern portion of the state. Again, using water quality data and the geographic locations of both water bodies and anglers, Jakus et al. (1997) estimated the economic impact of the fish consumption advisories. Anglers living in central Tennessee suffered a \$17.92 per trip per season loss from the advisories, and anglers in eastern Tennessee suffered a \$38.27 loss (1990 dollars). Therefore, considering an angler population of 146,450 individuals, the impact of this level of toxification into perpetuity, using a five percent discount rate, is approximately \$65.96 million.

These results indicate that fish advisories impose substantial economic cost on anglers in the United States. Measuring the marginal changes in toxification that would occur in the absence of the CAAA is not possible, but it is plausible to state that continued HAP emissions impose a cost on society if they result in the issuance of additional fish advisories. Any efforts to minimize these emissions, including the CAAA, may generate corresponding benefits.

If air deposition of toxics results in statewide fishing advisories (e.g., Connecticut, Washington D.C., Illinois, Maine, Massachusetts, Missouri, New Hampshire, New Jersey, New York, North Carolina, Ohio, Vermont), substitution away from recreational fishing for other activities may begin to occur. No models are available to estimate the economic impact of a large-scale substitution away from recreational fishing. The RUM approach does not adequately capture the magnitude of ubiquitous toxification because the models measure only the choice to participate in the activity and not the welfare implications of participation in alternative activities, nor do they account for the industries that provide supplies and services to anglers in the region. However, the economic cost of statewide advisories could be substantial.

Although Montgomery and Needelman (1997) and Jakus et al. (1997) examined only two areas of the country - New York State and part of Tennessee - their work demonstrates that HAP emissions have a measurable economic cost when the consequence of these emissions is the issuance of fish advisories for recreational fisheries. While it is not possible to measure the differences in HAP deposition and the marginal ecological impacts that will result from the CAAA, it is clear that continued emissions of HAPs will result in further toxification of aquatic resources, and reductions in HAP emissions may provide economic benefits.

The toxification of freshwater ecosystems in the United States by mercury and dioxins is a problem, and emissions of mercury and dioxins to the atmosphere contribute significantly to the problem. Quantifying the magnitude of ecosystem effects of air toxics deposition is not yet possible, but it is clear that the deposition of air toxics to some ecosystems, such as freshwater recreational fisheries, can result in measurable economic costs.

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[22] A perpetuity is a stream of benefits, accrued over an infinite time horizon. A simplified formula for calculating a perpetuity of equal benefits accrued annually, in which the first payment is received at the end of year one, the second payment is received at the end of year two, etc., is: (Nominal Value of Benefit) / (Discount Rate)

Caveats and Uncertainties

Because of limitations in the currently available data and models, a comprehensive quantitative analysis of the ecological benefits of reduced mercury and dioxin emissions for recreational fishing is not possible. However, such an analysis may be possible in the foreseeable future.

- The potential for mercury and dioxins to persist for long periods of time in the environment is a confounding factor in this analysis. Because these pollutants can persist in aquatic ecosystems for decades, even though the CAAA may reduce their emissions, it is possible that the status of toxified ecosystems may not be significantly affected during the time frame of the analysis (i.e., through 2010).

- In addition, the persistent nature of toxification presents challenges with respect to how benefits are discounted over time. In those cases where recovery from toxification will take a number of years, the benefits accrued by society will be diminished in terms of their present value. In other words, if all air emissions ceased, many fish consumption advisories would remain in place until the fisheries recovered. If this recovery period were to extend for several decades, the present value of economic benefits from the eventual retraction of advisories could be reduced dramatically. In a cost-benefit decision analysis, these benefits might not justify the costs of HAP regulations. In this case, an inter-generational benefits assessment, where discounting is not applied, would be required.

- The global nature of mercury pollution is another confounding factor Because a significant portion of mercury deposited within the U.S. comes from the global pool, a decrease in U.S. emissions may be offset by increases in emissions in other countries. If this should occur, it might be difficult to detect or predict actual changes in the toxicity of U.S. aquatic ecosystems, despite reductions in U.S. emissions.

- To quantitatively assess the effects of mercury and dioxin emission reductions on recreational fishing, more and better data and models are required. The most pressing research needs in this area are a model that can predict the national fate and transport of dioxin, and models that can, on a national scale, convert mercury and dioxin deposition quantities to amounts of the contaminants in fish. Data to verify these models is also highly desirable.

- Even if it were currently possible to perform the analysis discussed here, it would likely capture only a fraction of all the benefits attributable to CAAA-mandated HAP emissions reductions. The analysis focused entirely on two HAPs and on one endpoint. Neither the potential benefits of reductions in the emissions of other HAPs nor other endpoints were considered.

Conclusions and Implications

Our analysis has identified four major categories of air pollutants that affect ecological structure and function: sulfur compounds, nitrogen compounds, tropospheric ozone, and hazardous air pollutants. Each of these pollutants is scientifically documented as a cause of ecosystem degradation due to acute and chronic exposure. Sulfur and nitrogen compounds contribute to episodic and chronic acidification of aquatic and terrestrial ecosystems, while the chronic deposition of nitrogen compounds alone may cause harmful eutrophication to terrestrial and aquatic ecosystems. Tropospheric ozone disrupts the normal functioning of plants, leading to acute, visible damages to terrestrial ecosystems, and chronic exposure at levels that do not produce acute damages may result in reduced growth rates and eventually alter ecosystem nutrient cycling. Finally, hazardous air pollutants deposited across the landscape are accumulating in aquatic organisms and subsequently entering both aquatic and terrestrial foodchains. Though the ecological impacts are not fully understood, the long-term effects of introducing hazardous air pollutants to ecosystems may be slow to manifest and irreversible in nature.

Ecological effects can occur at different levels of biological organization. Most effects that are currently quantifiable are understood at the individual or population level, perhaps because of the feasiblity of conducting controlled experiments at this level. For example, research on the effects of ozone on timber began with experimental research on the response of seedlings and leaves of mature trees to elevated levels of ozone. Only recently have modeling efforts begun to consider interactions of factors at the community level, taking into account the dynamics of competitive relationships among tree and plant species. Experimental research continues to progress toward a better understanding of the full range of ecological impacts including effects at the ecosystem level. Continued consideration of these higher-order effects of pollutants on ecological systems is necessary for a more complete understanding of the benefits of pollution control.

Because the chronic ecological effects of air pollutants may be poorly understood, difficult to observe, or difficult to discern from other influences on dynamic ecosystems, our analysis focuses on acute or readily observable impacts. Disruptions that may seem inconsequential in the short-term, however, can have hidden, long-term effects through a series of interrelationships that can be difficult or impossible to observe, quantify, and model. This factor suggests that many of our qualitative and quantitative results may underestimate the overall, long-term effects of pollutants on ecological systems and resources.

Summary of Quantitative Results

Although the effects of air pollutants on ecological systems are likely to be widespread, many effects may be poorly understood and lack quantitative effects characterization methods and supporting data. In addition, many of our quantitative results reflect an incomplete geographic scope of analysis; for example, we generated monetized acidification results only for the Adirondacks region of New York State. As a result, quantitative results we generate for the purposes of estimating the benefits of the CAAA reflect only a small portion of the overall impacts of air pollution on ecological systems. Our quantitative overview of effects nevertheless suggests that the overall impacts of air pollution are far greater than those quantified.

Table E-27
Summary of Monetized Ecological Benefits (millions 1990$)

Description of Effect	Air Pollutant	Geographic Scale of Economic Estimate	Range of Annual Impact Estimates in 2010	Primary Central Estimate for 2010	Primary Central Cumulative Impact Estimate 1990-2010	Key Limitations
Freshwater acidification	Sulfur and nitrogen oxides	Regional (Adirondacks)	$12 to $88	$50	$260	- Captures only recreational fishing impact - Incomplete geographic coverage leads to underestimate of benefits
Reduced tree growth - Lost commercial timber	Ozone	National	$190 to $1000	$600	$1,900	- Uncertainties in stand-level response to ozone exposure - Uncertainty in future timber markets
TOTAL MONETIZED ECONOMIC BENEFIT			$200 to $1,100	$650	$2,200	- Partial estimate that omits major unquantifiable benefits categories; see text

Note: Estimates reflect only those benefits categories for which quantitative economic analysis was supported. A comprehensive total economic benefit estimate would likely greatly exceed the estimates in the table. Range of estimates for timber assessment is based on variation in annual point estimates for 2005 through 2010.

Despite these limitations, it is important to recognize the magnitude of the monetized ecological benefits that we could estimate and reflect those results in the overall estimates of benefits generated in the larger analysis. Table E-27 provides a tabular summary of the results documented earlier in this appendix. It is not possible to indicate the degree to which ecological benefits are underestimated, but considering the magnitude of benefits estimated for the select endpoints considered in our analysis, it is reasonable to conclude that a comprehensive benefits assessment would yield substantially greater total benefits estimates.

Recommendations for Future Research

Previous sections of this appendix have discussed several areas for future research related to the individual research and analytic efforts conducted. From a broader perspective, there are three key research needs to improve benefits assessments of this type:

- Exemplary assessments that incorporate a greater emphasis on ecosystem structure and function rather than specific service flows;
- Assessments with broader geographic coverage of impacts categories assessed in this report; and
- More sophisticated treatment of uncertainty and complexity, including careful consideration of the irreversibility of ecosystem impacts.

Assessing Changes in Ecosystem Structure and Funtion

A major limitation of our quantitative analysis is that by focusing on individual acute and chronic impacts it is possible to lose sight of ecosystem-level changes to structure and function. These ecosystem-level changes could eventually lead to large-scale impacts far greater in degree and geographic extent. Determining the appropriate ecological level of analysis is crucial to properly account for ecological benefits that may accrue from environmental regulations. While quantifying the decrease in impacts on species attributable to air pollutant control is analytically tractable, the impact of pollutant reductions on ecosystem structure and function may be a more appropriate measure that can be further explored in future analyses.

Changes in ecosystem structure and function may not be obvious to the lay person, and the ultimate effects of such changes in ecosystems are sometimes unpredictable in scale and nature. Ecosystems affected by humankind may respond in a discontinuous manner around critical thresholds that are boundaries between locally stable equilibria. Complexity in ecosystems prevents analysts from using linear methods to “add up” the discrete ecological effects of pollution. Understanding the complex cause and effect relationships between pollution and deterioration of ecosystem structure and function is fundamental to making adequate policy decisions that will protect ecological resources. The isolation of service flows may often imply an oversimplified cause and effect relationship between pollution and the provision of the service flow, when more often the service flow is affected by complex non-linear relationships that govern ecosystem structure and function. The result is that ecosystem impacts may not be adequately assessed by analyses that focus on specific service flows.

One potentially fruitful approach to assessing impacts on the ecosystem scale would be to more adequately model a wide range of ecosystem functions that do not necessarily contribute to human welfare. Assessments at the watershed scale might provide an appropriate level of detail to more adequately characterize some of these intermediate service flows. This type of research effort would require close cooperation between air pollution specialists, ecologists, and economists to be most useful within the context of benefit-cost analyses such as this one.

Broader Geographic Scale

Several of the ecological analyses conducted to support the first prospective section 812 report are limited by their partial geographic coverage. For example, while nitrogen deposition is an important contributor to eutrophication in a wide range of Eastern and Gulf Coast estuaries, resource, time, and data availability constraints, as well as limitations in our ability to reasonably apply an avoided cost approach, prevented EPA from conducting a national economic assessment for this category of impacts. In this and many other effects categories, extension of the methods applied here to new geographic areas could greatly enhance the comprehensiveness of the physical effects and economic impact estimates.

Alternative Treatment of Uncertainty

At present a variety of economic schools of thought are converging on quantitative analysis of environmental impacts that integrate uncertainty, irreversibility and ecological complexity. Efforts within the field of "ecological economics" to develop structured appraisals of uncertainty associated with environmental management and procedural rationale for decision making have yielded a variety of theoretical proposals. Drepper and Mansson (1993) argue that most aspects of uncertainty are compressed into the discount rate for policy analysis, resulting in the inappropriate use of a constant positive discount rate for environmental existence values. These existence values, they argue, may be more appropriately assigned negative discount rates. Faucheux and Munda (1997) advance a similar criticism of the unified discount rate and posit that a differentiated discount rate be applied to multiple aspects of a policy decision according to the implied uncertainty of each aspect. This quantitative approach evolves into a multi-criteria decision framework that departs from conventional cost-benefit analysis. Alternatively, Hinterberger and Wegner (1997) abandon quantitative analysis as a futile exercise due to ecosystem complexity in favor of simply applying the precautionary principal of reducing any and all environmental impacts that have uncertain outcomes.

In the resource economics literature, discussion of alternatives to cost-benefit analysis when the magnitude of benefits or costs are uncertain have focused on the concept of quasi-option value (see Freeman 1993 for a summary). The term was coined by Arrow and Fisher (1974) to describe the potential welfare gain of altering the timing of development/preservation decisions under uncertainty and when at least one of the choices involves an inrreversible commitment of resources (either spent or preserved). While much of the quasi-option value literature suggests that adopting this type of framework would lead to greater environmental protection, Freeman (1993) argues that it is also possible that the information gained by some incremental development of ecological resources might be the only way to reduce uncertainty and gain information about the magnitude of the trade-offs involved in preventing ecological exposures. It is nonetheless important to recognize that option and quasi-option value should not be considered as additional components of willingness-to-pay, but rather a value of altering decision making practices (e.g., the value of moving from a benefit-cost framework based on expected value to a framework that better considers the value of information gained over time and the irreversibility of certain effects).

The main implication of this body of work is that cost-benefit analysis may well underestimate the value of both the costs and benefits of uncertain, irreversible environmental outcomes from public policy. From the cost perspective, regulating a pollutant that may have no environmental consequence may cause economic losses that reduce unknown investment and growth opportunities in the future. From the benefits perspective, the value of preserving ecosystem integrity may include the mitigation of irreversible damage to a variety of service flows previously not associated with simplified dose-response relationships between pollution and ecosystems. Applications of these principles in economic assessments, including more rigorous assessments of option and quasi-option value, probabilistic analysis of multiple scenarios, and value of information approaches have the potential to greatly increase the utility of uncertain ecological assessment results for the purposes of making environmental policies.

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